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Information uncertainty influences conservation outcomes when prioritizing multi-action management efforts

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Abstract

1. In managing various threats to biodiversity, it is important to prioritize multiple management actions, and the levels of effort to apply. However, a spatial conservation prioritization framework that integrates these key aspects, and can be generalized, is still missing. Moreover, assessing the robustness of prioritization frameworks to uncertainty in species responses to management is critical to avoid misallocation of limited resources. Yet, the impact of information uncertainty on prioritization of management effort remains unknown.

2. We present an approach for prioritizing alternative levels of conservation management effort to multiple actions, based on the ecological responses of species to management. We estimated species responses through a structured email-based expert elicitation process, where we also captured the uncertainty in individual experts' assessments. We identified priority locations and associated level of management of effort of four actions to abate threats to freshwater-dependent fauna, using a northern Australia case study, and quantified sensitivity of the proposed solution to uncertainty in the answers of each individual expert.

3. Achievement of conservation targets for freshwater-dependent fauna in the Daly River catchment would require 9.4 Million AU$ per year, for a total of approximately 189 Million AU$ investment over 20 years. We suggest that this could be best achieved through a mix of aerial shooting of buffalos and pigs, riparian fencing and chemical spraying of weeds, applied at varying levels of management effort in key areas of the catchment.

4. Uncertainty in experts' estimation of species responses to threats causes 60% of the species to achieve 80% of their conservation targets, which was consistent across target levels.

5. *Synthesis and applications*. Our prioritization approach facilitates the planning of conservation management at fine spatial scales and is applicable to terrestrial, freshwater and marine realms. Plan implementation may require policy instruments ranging from landowner stewardship agreements, market-based mechanisms and low-intensity land use management schemes; to regulation of commercial activities within portions of marine protected areas. However, assessing plan sensitivity to uncertainty in species response to management and finding ways of dealing with it in the prioritization rather than ignoring it, as often done, remains vital for effective achievement of conservation objectives.

Keywords: priority threat management, spatial conservation prioritization, optimal resource allocation, freshwaters, northern Australia, conservation management, conservation planning

Introduction

Spatial conservation prioritization aims to identify optimal sets of sites, and conservation management actions to prescribe at those sites, to achieve a conservation objective for multiple species, within some defined constraints (e.g., cost of actions) (Possingham, Ball & Andelman 2000). This is achieved by accounting for the contribution of management at individual sites to the overall conservation objective (i.e., complementarity) (Moilanen, Wilson & Possingham 2009). Originally, spatial conservation prioritization problems aimed to select a set of sites for establishment of protected areas (Possingham, Ball & Andelman 2000). However, recently, the attention has shifted towards real-world prioritization problems, with multiple actions and levels of within-site management effort (Watts *et al.* 2009; Moilanen, Leathwick & Quinn 2011; Pouzols & Moilanen 2013). Species responses to actions, and to their levels of effort, are uncertain and the effectiveness of any prioritization solution must be assessed against variability in input information, to avoid allocating insufficient effort or wasting limited resources (Moilanen *et al.* 2006; Rondinini *et al.* 2006; McDonald-Madden, Baxter & Possingham 2008; McCarthy *et al.* 2011). Yet, a unifying approach capable of addressing simultaneously different real-world complexities (e.g., multiple actions and levels of effort), and quantifying the impact of information uncertainty on prioritization solutions, is still missing.

The level of management effort to be allocated to an action refers to a site-specific factor, such as the time frame over which to conduct a certain management action (e.g., surveillance/monitoring, patrolling to reduce poaching, active control of an invasive species), or the budget to invest in the action (Chades *et al.* 2008; Hauser & McCarthy 2009; Chauvenet *et al.* 2010; Auerbach, Tulloch & Possingham 2014). Identifying the optimal level of management effort to allocate to an action within a site maximises the biodiversity benefits for a given dollar invested (i.e., cost-effectiveness) (Hauser & McCarthy 2009; McCarthy *et* *al.* 2010). However, considering multiple levels of management effort represents a main challenge in conservation prioritization, as it increases rapidly the number of management options available for a site (e.g., multiple combinations of actions and levels at each site). Recently, Cattarino et al. (2016) showed the improvement in cost-effectiveness generated from prioritizing continuous levels of management effort to multiple actions. However, in order to isolate the effect of continuous responses, Cattarino et al. (2016) made several simplifying assumptions (e.g., constant costs and theoretical species responses), which reduced the degree to which the study results can be extrapolated to other settings. A generalizable, spatial conservation prioritization scheme which captures a range of real-world complexities (i.e., multiple actions and levels of management effort, real costs and species responses) could improve prioritization of conservation management effort but has yet to be developed and tested.

Prioritizing conservation actions and levels of effort requires information on the responses of species to the actions or the threats addressed by the actions. However, data on species responses to actions are often incomplete or have associated uncertainty (i.e., epistemic uncertainty) (Regan, Colyvan & Burgman 2002). Uncertainty can be expressed in the form of empirical error measurements around a nominal estimate (Moilanen *et al.* 2006), or as a range of plausible values around an expert's best answer, as estimated through an elicitation process (Burgman, Lindenmayer & Elith 2005; Martin *et al.* 2012). Despite the pervasiveness of uncertainty in conservation decision problems, prioritization studies often consider a single value (nominal or most likely) as true species response (Carwardine *et al.* 2012; Mills *et al.* 2012; Chades *et al.* 2015). However, if the true benefit a species accrued following implementation of a management action is lower than the species benefit used in the prioritization, the management decision may fail to achieve conservation objectives (McDonald-Madden *et al.* 2010). Conversely, if the true benefit is higher than the benefit used in the prioritization, the prescribed effort might be higher than needed to achieve objectives, resulting in low cost-effectiveness. Therefore, assessing how uncertainty in species responses affects achievement of conservation objectives is crucial for effective allocation of conservation management effort.

Here we developed a conservation prioritization approach which considers multiple actions, and multiple levels of effort to allocate to each action. Our study addresses two main questions. What is the spatial location and overall cost of priority management actions, and their level of allocated management effort, to achieve specific conservation objectives, when we assume that species responses are known with complete certainty? What is the impact of uncertainty in species response estimates, here parameterized using expert knowledge, on achievement of conservation objectives? We answered those questions using a case study from northern Australia where we prioritized spatial allocation of four management actions, at varying levels of effort, to address threats to freshwater-dependent fauna.

Materials and Methods

Conceptual framework

We built on the problem addressed by Cattarino et al. (2016), who prioritized the allocation of alternative levels of management effort to multiple conservation actions, across multiple sites (planning units), to remediate threats to multiple species for the least cost, based on the ecological responses of species to actions (see Supporting Information). We expanded this problem to account for the varying intensity (i.e., magnitude) of a threat in a planning unit (e.g., spatial extent of a conflicting land use), which influences the level of management effort required to remediate the threat (Adams & Setterfield 2015). We collated information on species and threat intensity distribution and assumed that the intensity of a threat in a planning unit falls into one of three categories ('Low', 'Medium' and 'High') (Fig 1a-b). Species responses represent how the probability of persistence of the species varies under increasing threat intensity (Fig 1c). We quantified species responses and uncertainty around response information using expert elicitation. Information uncertainty was expressed as a range of values (*lower bound and upper bound*) around a most likely answer (*best guess*) (Fig 1c). Action costs were sourced from previously published studies (Fig 1d).

We developed a prioritization framework in which three potential levels of effort ('Low', 'Medium' and 'High') could be allocated to each action. Low level maintains (i.e., avoid increasing) the initial threat intensity, while Medium and High levels reduce threat category by one and two categories, respectively (Fig 1e). We assumed that the cost of actions increased linearly with the level of management effort (Fig 1f). The aim of the framework is to identify a set of priority areas, the type of management action, and level of effort prescribed within those areas, to achieve a minimum representation for each species, while minimizing management costs (Fig 1g-i). The minimum representation, or *target*, is the area of occupancy of each species, which is expressed as the product of the probability of persistence, achieved through selected actions and effort, and the area of occupancy of a species in the planning units where actions and effort are selected (Fig 1h).

To evaluate the effect of information uncertainty on achievement of conservation objectives, we first generated a prioritization solution using expert best guesses, which is analogous to assuming no uncertainty around experts' most likely answers (Fig 1g). This reflects a common assumption in conservation planning. Then, in post-hoc analysis, we quantified species representation (in the prioritization solution) by assuming that experts' answers were uncertain. We simulated the effect of experts' uncertainty by using the lower and upper bounds of the experts' answers (averaged across all experts) as estimates of species responses (Fig 1l). This reflects the extent to which the implementation of a conservation plan may result in target shortfalls (in the case that the true species response is the lower bound) or in an unexpected windfall (in the case that the true response is the upper bound).

Study area and species

The Daly River catchment is in tropical northern Australia and extends over 53,000 km^2 of tropical savannah woodland (Chan *et al.* 2012). Despite low levels of clearing (~5%) and existing conservation areas (~10%), long-term persistence of freshwater dependent fauna in the Daly River catchment is affected by major threatening processes, including invasive animals, agricultural land use (particularly grazing, which is the dominant land use) and proliferation of aquatic weeds (Adams *et al.* 2014).

We defined a spatial framework consisting of 865 hydrologically-defined sub-catchments, which represent the planning units of analysis (see Supporting Information, section 1). We considered a suite of freshwater-dependent taxa (44 fishes, 8 freshwater turtles and 86 water birds) and four major threats to freshwater biodiversity: (1) introduced water buffalos (*Bubalus bubalis*); (2) feral pigs (*Sus scrofa*); (3) grazing land use; and (4) para grass (*Brachiaria mutica*) - a highly invasive weed.

Management actions and costs

For each threat we considered a remediating management action: aerial shooting of water buffalo and feral pigs, building cattle-proof fences along riparian zones to reduce cattle trampling and other damages to freshwater ecosystems, and chemical spraying of para grass (Bayliss & Yeomans 1989; Carwardine *et al.* 2011; Setterfield *et al.* 2013). The management costs of implementing different levels of effort of each action (AU$/ha/year) were sourced from peer-reviewed studies in other river-floodplain ecosystems of northern Australia (see

Supporting Information, section 3.4). Cost estimates were calculated as the net present value of the sum of set-up costs (long-term capital, materials, supplies and labour) in the first year and the ongoing annual maintenance costs, assuming an ongoing investment in the action over 20 years and a 5% discount rate (Carwardine *et al.* 2012). We assumed that the cost of implementing increasing levels of management effort to an action in each planning unit increased as a linear function of the level of allocated effort (Santika *et al.* 2015). When costing the actions, we assumed that: (1) aerial shooting was conducted over the area of the entire planning unit; (2) riparian fencing was implemented along the stream network within each planning unit; and (3) para grass spraying occurred over the estimated extent of para grass infestation in each planning unit.

Species' ecological responses to threats

We asked experts to estimate the ecological responses of species to threats using a structured, email-based elicitation approach (McBride *et al.* 2012) (see Supporting Information, section 2). We categorized species from all faunal groups into 18 different ecological groups, based on similarities in ecological requirements and behaviour (see Table S1.1). We then engaged 13 experts in the ecology and conservation of freshwater fishes, turtles and/or water birds via email and asked them to estimate probabilities of persistence of species in different ecological groups, given exposure to three intensities of each threat (Carwardine *et al.* 2012). Following a 4-point elicitation procedure, we asked the experts to provide the most likely value (best guess), lowest and highest plausible values, and level of confidence they had that the true value of persistence lay within the lowest-highest value bound (Speirs-Bridge *et al.* 2010). This interval represents the uncertainty of one expert in the actual response value exhibited by a taxon. We collected a total of 72 ecological responses (18 ecological groups × 4 threats) (Fig. S3), which were then used in the prioritization.

Benefits of actions

The benefit of a particular level of effort, was equal to the increase in probability of persistence following action implementation. For example, given initial 'High' threat intensity, implementing a 'Medium' level of effort for an action reduces the threat to 'Medium' intensity and therefore the benefit corresponded to the species persistence under 'Medium' intensity of that threat (the persistence value is averaged across experts) (see Supporting Information, section 3.3). The benefit of the 'Low' effort corresponded to the probability of persistence under the initial intensity of the threat.

Optimization approach

We used the optimization approach described in Cattarino et al. (2016), which is based on simulated annealing, to find a near-optimal solution to the action prioritization problem (see Supporting Information, sections 3.2 and 3.5). This approach is similar to the one adopted by the spatial conservation prioritization software, Marxan (Ball, Possingham & Watts 2009). However, while Marxan focuses on planning unit selection, our optimization algorithm iteratively removes from, or adds to, the solution, one level of management effort for one action in one planning unit, at a time.

Quantifying the effect of information uncertainty

To assess the effect of expert uncertainty on achievement of conservation targets, we adopted a two-stage procedure. We first identified the set of priorities (sites, actions and levels of effort) to assess a conservation target for each species, using the expert best guesses (averaged across experts) as the 'true' response. These spatial priorities reflect real-world cases in which managers use best available information to set priorities, ignoring uncertainty around species response. We then estimated the extent to which implementing these sets of priorities may result in either over- or under-achievement of species targets, when the true species responses deviate from expert best guesses. Upper and lower estimates of species representation were calculated by assuming that the true response corresponded to either the lower bound or the upper bound of the experts' answers (averaged across experts), respectively.

We calculated the percentage change in the representation of species *j* under different assumptions of true response (*observed* representation), relative to the representation level achieved in the original prioritization (*expected* representation). Percentage changes were calculated as:

$$\Delta_j = \frac{O(R)_j - E(R)_j}{E(R)_j} \times 100 \qquad \text{eqn 3}$$

where $E(R)_j$ and $O(R)_j$ are the expected and observed representation for species *j*, respectively. A positive change occurs when a species is less sensitive to the threat than expected, because for a given action-effort combination, the true species probability of persistence is higher than what was assumed in the prioritization. The consequence of this is that, when a manager uses the results obtained using the best guess prioritisation, the effort selected in the prioritization is higher than what is actually needed. This means that the solution is less cost effective, but will assure achievement of targets.

A negative change occurs when a species is more sensitive to the threat than expected, because for a given action-effort combination, the species probability of persistence is lower than what was assumed in the prioritization. Therefore, the effort selected in the prioritization is lower than the effort needed to achieve the targets. Consequently, we might fail to achieve species targets.

Analysis

We applied species targets proportional to each species area of occupancy to ensure representing the whole distribution of rare species and avoid over representing common ones (Rodrigues *et al.* 2004). We set a fixed target corresponding to 100% of the range of species with an area of occupancy smaller than 500 km^2 (20% of species). We also set a fixed target of 10% of the range of species with an area of occupancy larger than 10,000 km^2 (24% of species). The target for species with area of occupancy of intermediate size (57% of species), was calculated using linear interpolation (see Supporting Information, section 1). We investigated how sensitive our results were to species targets and repeated the analysis for a range of target level (see Supporting Information, section 5.3).

Results

Cost and spatial priorities of conservation management effort in the Daly River catchment

Ensuring ecological persistence of freshwater-dependent fauna in the Daly River catchment would cost approximately 9.4 Million AU$ per year, for a total of just below 189 Million AU$ investment over 20 years (Table 1). This requires, for example, conducting low levels of aerial shooting of water buffalos over around 17,700 km^2, and medium levels of aerial shooting of feral pigs over 7,000 km^2, per year (Table 1). Long-term persistence of freshwater dependent fauna in the Daly also requires riparian fencing at low and medium levels of management effort, over 142 and 13 km^2 of stream area, respectively, and low, medium and high levels of chemical spraying over 207, 225 and 562 km^2, respectively, of para grass infestation, per year (Table 1). Priority areas selected for allocation of conservation management effort are mainly located on the floodplain and tributary streams of the lower Daly River catchment, along the main stem of the Daly River, and in the headwaters in the north-eastern part of the catchment (Fig. 2).

Effect of an expert's own confidence level on target achievement

When the lower bounds of experts' answers were assumed to be the true species responses, average species representation was 20% lower than when the experts' best guesses were used as true responses (Fig 3). This pattern was consistent across different target levels (Figure S7). The observed drop in species representation was due to almost 60% of the species achieving approximately 80% of their conservation targets (Fig. S5-S6).

When the upper bounds of experts' answers were assumed to be the true species responses, average species representation was 2.5% higher than when the experts' best guesses were used as true responses (Fig 3). Positive change in species representation did not translate into an increasing number of species represented above target levels. This is because most of the species were already represented at or above target level when using the experts' best guesses (Figure S5).

Discussion

We have developed a novel approach for optimizing the spatial allocation of priority threat management effort (Moilanen, Wilson & Possingham 2009; Carwardine *et al.* 2012; Chades *et al.* 2015). We demonstrated the efficacy of our approach using a case study from northern Australia and identified priority areas where a mix of aerial shooting of water buffalos and feral pigs, riparian fencing and chemical spraying of para grass, applied at varying degrees of management effort, are needed to conserve freshwater biodiversity. We also showed that, in a consistent fashion across a range of conservation objectives, uncertainty in estimates of species response to actions undermines the capacity to achieve conservation objectives. This suggests that using experts' best answers for conservation decisions may increase the risk of misallocating limited conservation resources or lead to overoptimistic assessment of conservation progresses, as species are considered able to persist in the face of threats, when in reality they are not. Our approach can aid planning conservation management strategies at fine spatial scales. Our findings call for improving accuracy of experts' answers to be used in prioritization and highlights the importance of assessing the performance of prioritization plans based on experts' best answers.

We identified key areas for allocation of threat-specific management efforts in the Daly River catchment, with the Anson Bay floodplain (north-east), the Daly River Middle Reaches (central) and the Arnhem Land Plateau / Katherine River headwaters (north-west) being top priorities (Figure 2). These areas have been identified by the Northern Territory government as sites of conservation significance and feature among the top conservation priorities of previous freshwater prioritization studies in the region (Linke *et al.* 2012; Northern Territory Government 2017). However, previous studies assumed that priority areas were converted into protected areas by acquiring their land, which is often not a viable conservation strategy due to the pressure imposed by other human activities (e.g. agriculture). In contrast, we identified the specific management actions (and the required level of management effort) to undertake in priority areas. While invasive herbivore and aquatic weed control are priority actions in the coastal floodplain and Arnhem plateau, riparian fencing was selected for most of the Daly middle reaches. Furthermore, these areas should be allocated a higher level of management effort (e.g. spatial extent or hours of management) relative to other areas, according to our analysis. This information provides much more operational detail for protected area managers and land owners confronted with threat management.

Our study highlights the importance of the spatial distribution of threats and species responses to the associated actions in driving spatial allocation of conservation management effort (Tulloch *et al.* 2015). Among the spatial priorities we identified the south-western portion of the catchment (Katherine River's main channel) as a key area where to undertake riparian fencing. This area overlaps with the spatial distribution of cattle grazing, a threat to which the species considered here (e.g. fishes) are particularly vulnerable (Figure S3). The Kathrine River's main channel however was missing from the priority areas identified by previous study in the region, which was largely based on the same set of species considered here, but did not account for the spatial distribution of threats and the responses of species to the remediating actions (Linke *et al.* 2012).

The cost of threat-specific conservation actions (e.g. invasive species management) is comparable to those estimated by other studies in the Daly and in other parts of northern Australia (Kimberley) (Adams, Pressey & Stoeckl 2012; Carwardine *et al.* 2012). However, our total costs estimates differ from those of other similar studies. This is unsurprising given the differences in threats and management actions considered. In addition, we assume that prescribed management occurred in the portion of the planning units where the threats occurred, as opposed to managing the entire planning unit, as assumed in other studies. This means that it is hard to make meaningful cost comparisons across studies. Furthermore, our cost estimates should be interpreted conservatively, as we extrapolated them from studies conducted at different spatial scales and did not account for the cost reduction obtained by managing large areas of land. Accounting for such economies of scale would have likely resulted in generating even lower cost estimates, thus increasing cost-efficiency of our approach (Armsworth *et al.* 2011).

Our findings suggest that considering variability in species responses to actions when prioritizing conservation management effort should become best practice, if we want to minimize the risk of undermine conservation objectives. Unfortunately, current approaches often do not quantify the effect of response uncertainty, but rather tend to make use of best (most likely) responses estimates, which might not represent true species responses (Carwardine *et al.* 2012; Mills *et al.* 2012; Chades *et al.* 2015). We showed that ignoring the variability in an expert's own range of potential answers (the confidence level) might lead to failure to achieve conservation targets. Our result highlights the importance of considering uncertainty from the onset of the planning stage. Doing so might reduce the risk of generating solutions highly susceptible to uncertainty in expert knowledge. This is particularly relevant when specific conservation objectives must be met, as we have demonstrated in our analysis. One way to properly account for uncertainty in species responses when prioritization actions is to create "robust" spatial prioritization solutions, which can tolerate large variations in the expected response values, without compromising the achievement of conservation targets (Moilanen *et al.* 2006; Burgman *et al.* 2010). Previous studies have addressed this in the context of conservation problems with only site reservation action and could be expanded by incorporating multiple actions.

Study limitations

We assumed categorical levels of threats and management effort, as well as a linear relationship between level of management effort and amount of threat reduced. However, the use of continuous response curves or varying shapes is likely to provide the greatest cost-efficiency (Cattarino *et al.* 2016). Nevertheless, for the current applied study, we decided to use categorical responses, which facilitated experts' task of estimating species responses, by providing a benchmark, when available, for each individual threat intensity (e.g., estimated number of buffalos/km^2). Moreover, defining more detailed categories would have required more precise threat distribution information, particularly on the intensity of individual threats, which was unavailable. Our framework can be easily expanded to incorporate continuous responses of different shapes. Where and when finer-scale information on threat intensity are available, an infinitesimal (continuous) number of threat and effort categories can be implemented. Moreover, if the relationship between amount of effort and threat intensity is known, or can be easily elicited from experts, the present framework can be broadened to incorporate different relationships between threat and effort categories, or curves of varying shapes.

Effective on ground management is not purely a matter of mathematical optimization - there are other social and human factors which have not been considered here. For example, we did not consider landowners' willingness to engage in conservation practices, such as aquatic weed control, river bank restoration and improved invasive herbivore management (Honig *et al.* 2015). Given most of the Daly River catchment is privately owned or managed, failing to account for landowners' willingness to participate into conservation programs, or more generally their attitudes towards conservation, is likely to represent a barrier to achievement of conservation objectives.

Management implications

Our approach can aid local and regional government agencies to plan and implement priority threat management at fine spatial scales (Wilson *et al.* 2011; Carwardine *et al.* 2012; Game, Kareiva & Possingham 2013). This may require putting in place policy instruments which target individual properties, such as landowner stewardship agreements, where portions of the property are set aside for conservation or where best farming management practices are adopted (Claassen, Cattaneo & Johansson 2008; Moon & Cocklin 2011). Such agreements might be directed at implementing specific actions at specific levels of effort, such as removing a number of invasive herbivores per km^2, setting up cattle fences of specific length along a river to keep cattle away, or applying chemical spraying to a portion of weed-infested floodplain. Alternative policy tools include market-based mechanisms, such as labelling or certification of products (e.g. beef) produced on land where conservation practices are adopted, and land use management schemes promoting sharing land between conservation and agricultural production, through adoption of less intensive farming practices (e.g. rotational grazing) (Fischer *et al.* 2008; Higgins, Dibden & Cocklin 2008). Our approach may help prioritizing fine-scale management in settings different from freshwater/terrestrial ones. In marine settings, for example, management regimes which can be prioritized with our approach include exclusion of fishing and other economic activities around sensitive sites (e.g. coral reef), restoration of specific tracts of coastal habitat (e.g. mangroves), and regulation of fishing pressure within portions of a marine protected area (Foley *et al.* 2010; Adame *et al.* 2015; Costello *et al.* 2016).

The marked effect of information uncertainty on achievement of conservation objectives highlights the importance of improving accuracy of species response estimates to management. This may require to refine the expert elicitation process by, for instance, giving experts opportunities to discuss their answers in person and testing the experts beforehand with similar questions to the ones they will be required to answer (Burgman *et al.* 2011; Hemming *et al.* 2018). However, insufficient resources (time and money) often require managers to make quick decisions with the best available information, i.e. experts' best guesses. In this case, we recommend to carefully assess the performance of the conservation plan developed using the best response estimates (Sarkar *et al.* 2006). If the plan turns out to perform poorly (some species in decline), a manager might need to calibrate management effort in selected priority areas. Our approach represents a flexible tool which may aid managers with effort calibration, as it prioritizes variable levels of management effort at individual sites. Varying the level of effort to apply to actions in existing priority areas might be more cost-effective than finding new priority areas given resources (e.g., personnel and vehicles) are already deployed on site.

Authors' contribution

LC performed the analysis and drafted the manuscript. All authors contributed critically to the drafts and gave final approval for publication.

Data accessibility

Data and R code are available from the Zenodo digital repository.
DOI:10.5281/zenodo.1184586 (Cattarino *et al.* 2018).

Supporting Information

Species and threats considered in the study (Section 1), Expert elicitation (Section 2), Prioritization (Section 3), and Effect of expert knowledge uncertainty on target achievement (Section 4) are available online. The authors are solely responsible for the content and functionality of these materials. Queries (other than absence of the material) should be directed to the corresponding author.

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Table 1. Average cost per year, over 20 years and spatial extent of prescription application, of different levels of management effort, for different actions prescribed in the Daly River catchment.

Action	Management effort	Annual cost	20-year cost	Treated area *
Aerial shooting of buffalos	Low	1.10	21.96	17,713
Aerial shooting of buffalos	Medium	0.17	3.42	2,762
Aerial shooting of buffalos	High	-	-	-
Aerial shooting of pigs	Low	1.49	29.80	9,142
Aerial shooting of pigs	Medium	1.16	23.10	7,086
Aerial shooting of pigs	High	-	-	-
Riparian fencing	Low	2.74	54.82	142
Riparian fencing	Medium	2.10	42.08	13
Riparian fencing	High	-	-	-
Chemical spraying	Low	0.19	3.72	207
Chemical spraying	Medium	0.17	3.40	225
Chemical spraying	High	0.32	6.35	562
Total		9.43	188.66	37,853

*area (km^2) of the planning units where each level of effort (for different actions) is prescribed. For aerial shooting of buffalos and pigs, treated area is the area of the planning units; (2) for riparian fencing, treated area is the stream area (assuming a 100 m river width) in the planning units; (3) for chemical spraying of para grass, treated area is area of the planning units infested with Para grass.

Figure 1. Conceptual framework of the study. We assembled input data on spatial distribution of species (a), intensity of different threats (b), species responses (c) and cost of different remediating actions (d). Species responses were estimated using expert elicitation as a best guess and an upper-lower bound interval, which represented uncertainty in an expert's answer. We assume that different levels of management effort could be allocated to each remediating action to reduce threat intensity and improve species persistence (P_H, P_M and P_L) (e). The cost of implementing management effort to an action in a planning unit increases as a linear function of the level of allocated effort (f). To evaluate the effect of information uncertainty on target achievement, we first assumed that species responses were known without uncertainty and used the experts' best guesses (averaged across experts) as estimates of species responses to achieve representation targets in the prioritization (g-i). We then quantified the impact of uncertainty around species response estimates on target achievement, by re-calculating species representation in the prioritization solution using the experts' lower and upper bounds estimates of species responses (averaged across experts) (h and l). Representation, *R*, of a species in a planning unit equals the sum of the probability of persistence (*P*) achieved through implementation of different actions, multiplied by the area of occupancy (*a*) of the species in the planning unit.

Figure 2. Spatial distribution of management effort allocated to four different actions in the Daly River catchment. Results are shown for the best solution of 10 replicates and best guess expert estimate (averaged across experts).

Figure 3. Percentage change in species representation, relative to the best guess scenario, when the true species responses deviate from an expert best guess, due to an expert's own confidence level. The value on the *y* axis is the percentage change value averaged across species (± 1 SE). Best guess scenario refers to when the best guesses of individual experts (averaged across experts) were assumed to be the true species responses (continuous 0 line). The effect of an expert's own confidence level refers to when the lower and upper bounds of individual experts (averaged across experts) were assumed to be the true species responses. Displayed values are from the run with the lowest objective function, among a set of 10 replicate runs.

Supporting Information

1. Species and threats considered in the study

1.1 Species data

We considered 138 species from three different freshwater-dependent faunal groups (fishes, turtles and waterbirds) (Table S1). Species in each faunal group were classified among 6 different ecological groups on the basis of similar ecological traits (habitat and food requirements). We sourced the spatial distribution of 44 freshwater fish species, 8 freshwater turtle species, and 86 waterbird species, from a larger database on aquatic species distributions in northern Australia (Kennard 2010). Species distribution was modelled using a data set of 1,328 presence-absence points for fishes, 2,109 presence-only points for waterbirds and 350 presence-only points for turtles (Kennard 2010; Hermoso, Kennard & Linke 2012). Using a set of climatic, topological and environmental variables (vegetation, hydrology, primary productivity) as well as a variable measuring the degree of river flow alteration, observation data were fitted to Multivariate Adaptive Regression Splines (MARS) models. Model output was predicted probability of occurrence for each species at fine spatial scale (average area of predictive polygons was 3.6 km^2 for fish and 72 km^2 for waterbirds, respectively).

When modelling species distribution, it is important to feed into the model, together with presence data, information on where the species is absent, to produce unbiased estimates of probability of occurrence (e.g. avoiding predicting species occurrence in areas where the species does not occur, or not predicting species occurrence in areas where the species does occur) (Zaniewski, Lehmann & Overton 2002). To avoid generating biased predictions, in the case of faunal groups for which absence records were not available (waterbirds and turtles), "inventory" pseudoabsences were selected from the presence-only datasets. Briefly, if a site was visited and the species was not found, the site was treated as an absence record. The use of inventory pseudoabsences in MARS model has been shown to outperform other approaches to pseudoabsence selection, such as random sampling across the entire environmental space (Elith & Leathwick 2007).

Predicted probability of occurrence was converted to presence absence using a threshold approach (Kennard 2010; Hermoso, Kennard & Linke 2012). By applying a cut-off along the continuum of values of predicted probability of occurrence, this approach allows to distinguish between areas where a species is present from areas where a species is absent. By doing it is possible to convert a probabilistic measure of habitat suitability into a binary measure of species distribution, as routinely done in conservation planning studies (Guisan & Thuiller 2005; Wilson *et al.* 2005). We calculated the total area of occupancy of each species in each one of the 865 planning units of our study, by summing the area of the original predictive polygons where the species was assumed to be present, within each of the 865 planning units (Table S1).

Table S1. Faunal group, ecological group, name and area of occupancy of all species considered in the prioritization.

Faunal and ecological group	Genus	Species	Common name	Area of occupancy (km^2)	Target (km^2)
Fishes					
Large-bodied migratory carnivore	*Neoarius*	*berneyi*	Berney's catfish	69	69
Large-bodied migratory carnivore	*Neoarius*	*graeffei*	Lesser salmon catfish	1,432	1,306
Large-bodied migratory carnivore	*Neoarius*	*leptaspis*	Triangular shield catfish	399	399
Large-bodied migratory carnivore	*Neoarius*	*midgleyi*	Shovel-nosed catfish	272	272
Large-bodied migratory carnivore	*Lates*	*calcarifer*	Barramundi	7,167	2,640
Large-bodied migratory carnivore	*Megalops*	*cyprinoides*	Tarpon	5,620	2,894
Large-bodied herbivore/omnivore	*Nematalosa*	*erebi*	Bony bream	24,681	2,468
Large-bodied herbivore/omnivore	*Arramphus*	*sclerolepis*	Snub-nosed garfish	98	98
Large-bodied herbivore/omnivore	*Liza*	*ordensis*	Ord River mullet	698	685
Large-bodied carnivore	*Anodontiglanis*	*dahli*	Toothless catfish	1,517	1,371
Large-bodied carnivore	*Neosilurus*	*ater*	Narrow-fronted catfish	8,445	2,089
Large-bodied carnivore	*Neosilurus*	*hyrtlii*	Hyrtl's tandan	44,858	4,486
Large-bodied carnivore	*Neosilurus*	*pseudospinosus*	False-spined catfish	157	157
Large-bodied carnivore	*Strongylura*	*krefftii*	Freshwater longtom	11,876	1,188
Large-bodied carnivore	*Ophisternon*	*sp*	Swamp eel	5	5
Large-bodied carnivore	*Toxotes*	*chatareus*	Seven-spot archerfish	9,444	1,442
Large-bodied carnivore	*Toxotes*	*lorentzi*	Primitive archerfish	560	556
Large-bodied carnivore	*Oxyeleotris*	*lineolatus*	Sleepy cod	2,079	1,768
Large-bodied carnivore	*Oxyeleotris*	*selheimi*	Giant gudgeon	4,499	2,795
Small-bodied migratory invertivore	*Glossogobius*	*aureus*	Golden goby	1,763	1,552
Small-bodied migratory invertivore	*Glossogobius*	*giurus*	Flathead goby	4,258	2,742
Small-bodied migratory invertivore	*Glossogobius*	sp2MUNROI	Munro's goby	8	8
Small-bodied migratory invertivore	*Hypseleotris*	*compressa*	Empire gudgeon	199	199
Small-bodied migratory invertivore	*Leptachirus*	*triramus*	Freshwater sole	41	41
Grunters	*Amniataba*	*percoides*	Barred grunter	26,516	2,652
Grunters	*Hephaestus*	*fuliginosus*	Black bream	5,547	2,895
Grunters	*Leiopotherapon*	*unicolor*	Spangled perch	52,124	5,212

Grunters	*Syncomystes*	*butleri*	Butler's grunter	1,118	1,053
Small-bodied invertivore	*Porochilus*	*rendahli*	Rendahl's catfish	705	691
Small-bodied invertivore	*Melanotaenia*	*nigrans*	Black-banded rainbowfish	2,156	1,818
Small-bodied invertivore	*Pseudomugil*	*tennellus*	Delicate blue-eye	18	18
Small-bodied invertivore	*Denariusa*	*bandata*	Pennyfish	227	227
Small-bodied invertivore	*Craterocephalus*	*marianae*	Mariana's hardyhead	46	46
Small-bodied invertivore	*Melanotaenia*	*exquisita*	Exquisite rainbowfish	8,068	2,284
Small-bodied invertivore	*Pingalla*	*midgleyi*	Midgley's grunter	508	507
Small-bodied invertivore	*Hypseleotris*	*burrawayi*	Barraway's carp gudgeon	72	72
Small-bodied invertivore	*Craterocephalus*	*stercusmuscarum*	Fly-specked hardyhead	12,277	1,228
Small-bodied invertivore	*Craterocephalus*	*stramineus*	Strawman, Blackmast	2,737	2,157
Small-bodied invertivore	*Melanotaenia*	*australis*	Western rainbowfish	46,845	4,684
Small-bodied invertivore	*Ambassis*	spNORTHWEST	Northwest glassfish	7,112	2,657
Small-bodied invertivore	*Ambassis*	*agrammus*	Sailfin glassfish	8,565	2,021
Small-bodied invertivore	*Ambassis*	*macleayi*	Macleay's glassfish	3,028	2,303
Small-bodied invertivore	*Glossamia*	*aprion*	Mouth almighty	16,410	1,641
Small-bodied invertivore	*Mogurnda*	*mogurnda*	Northern trout gudgeon	31,293	3,129
Turtles				2,268	1,888
Pig-nosed turtle	*Carettochelys*	*insculpta*	Pig-nosed turtle	14,760	1,476
Sandstone snake-necked turtle	*Chelodina*	*burrungandjii*	Sandstone snake-necked turtle	12,710	1,271
Northern snake-necked turtle	*Chelodina*	*rugosa*	Northern snake-necked turtle	29,743	2,974
Northern snapping turtle	*Elseya*	*dentata dentata*	Northern snapping turtle	463	463
Common sawshell turtle	*Myuchelys*	*latisternum*	Common sawshell turtle	8,528	2,042
Short-necked turtles	*Emydura*	*victoriae*	Northern red-faced turtle	61	61
Short-necked turtles	*Emydura*	*tanybaraga*	Northern yellow-faced turtle	38,777	3,878
Short-necked turtles	*Emydura*	*subglobosa worrelli*	Diamondhead or Worrell's turtle	5,641	2,894
Waterbirds				8,151	2,243
Ducks, small grebes and Jacana	*Anas*	*gracilis*	Grey Teal	3,199	2,381
Ducks, small grebes and Jacana	*Anas*	*superciliosa*	Pacific Black Duck	10,173	1,017
Ducks, small grebes and Jacana	*Aythya*	*australis*	Hardhead	3,775	2,604

Ducks, small grebes and Jacana	*Dendrocygna*	*arcuata*	Wandering Whistling-Duck	11,681	1,168
Ducks, small grebes and Jacana	*Malacorhynchus*	*membranaceus*	Pink-eared Duck	10,304	1,030
Ducks, small grebes and Jacana	*Nettapus*	*pulchellus*	Green Pygmy-Goose	4,719	2,833
Ducks, small grebes and Jacana	*Tadorna*	*radjah*	Radjah Shelduck	1,003	955
Ducks, small grebes and Jacana	*Irediparra*	*gallinacea*	Comb-crested Jacana	6,986	2,693
Ducks, small grebes and Jacana	*Poliocephalus*	*poliocephalus*	Hoary-headed Grebe	1,286	1,191
Ducks, small grebes and Jacana	*Tachybaptus*	*novaehollandiae*	Australasian Grebe	9,295	1,550
Herbivores	*Cygnus*	*atratus*	Black Swan	10,444	1,044
Herbivores	*Dendrocygna*	*eytoni*	Plumed Whistling-Duck	2,853	2,217
Herbivores	*Anseranas*	*semipalmata*	Magpie Goose	584	580
Herbivores	*Fulica*	*atra*	Eurasian Coot	2,483	2,017
Herbivores	*Gallinula*	*ventralis*	Black-tailed Native-hen	2,881	2,231
Herbivores	*Porphyrio*	*porphyrio*	Purple Swamphen	986	941
Large wading birds	*Esacus*	*neglectus*	Beach Stone-curlew	535	533
Large wading birds	*Haematopus*	*fuliginosus*	Sooty Oystercatcher	27,186	2,719
Large wading birds	*Haematopus*	*longirostris*	Pied Oystercatcher	11,006	1,101
Large wading birds	*Ardea*	*alba*	Great Egret	32,575	3,258
Large wading birds	*Ardea*	*intermedia*	Intermediate Egret	5,099	2,877
Large wading birds	*Ardea*	*pacifica*	White-necked Heron	11,489	1,149
Large wading birds	*Ardea*	*picata*	Pied Heron	15,913	1,591
Large wading birds	*Ardea*	*sumatrana*	Great-billed Heron	1,709	1,513
Large wading birds	*Bubulcus*	*ibis*	Cattle Egret	9,335	1,522
Large wading birds	*Butorides*	*striatus*	Striated Heron	34,021	3,402
Large wading birds	*Egretta*	*garzetta*	Little Egret	43,004	4,300
Large wading birds	*Egretta*	*novaehollandiae*	White-faced Heron	44,089	4,409
Large wading birds	*Ixobrychus*	*flavicollis*	Black Bittern	6,936	2,707
Large wading birds	*Nycticorax*	*caledonicus*	Nankeen Night Heron	4,886	2,856
Large wading birds	*Ephippiorhynchus*	*asiaticus*	Black-necked Stork	9,810	1,158
Large wading birds	*Platalea*	*flavipes*	Yellow-billed Spoonbill	6,597	2,786
Large wading birds	*Platalea*	*regia*	Royal Spoonbill	10,208	1,021
Large wading birds	*Plegadis*	*falcinellus*	Glossy Ibis	16,442	1,644
Large wading birds	*Threskiornis*	*molucca*	Australian White Ibis	11,890	1,189

Large wading birds	*Threskiornis*	*spinicollis*	Straw-necked Ibis	574	570
Large wading birds	*Grus*	*rubicunda*	Brolga	792	770
Small wading birds and shorebirds	*Charadrius*	*leschenaultii*	Greater Sand-plover	596	590
Small wading birds and shorebirds	*Charadrius*	*mongolus*	Lesser Sand-plover	2,211	1,852
Small wading birds and shorebirds	*Charadrius*	*ruficapillus*	Red-capped Plover	2,840	2,211
Small wading birds and shorebirds	*Charadrius*	*veredus*	Oriental Plover	1,382	1,266
Small wading birds and shorebirds	*Erythrogonys*	*cinctus*	Red-kneed Dotterel	344	344
Small wading birds and shorebirds	*Pluvialis*	*fulva*	Pacific Golden Plover	14,825	1,483
Small wading birds and shorebirds	*Pluvialis*	*squatarola*	Grey Plover	3,883	2,639
Small wading birds and shorebirds	*Vanellus*	*miles*	Masked Lapwing	6,845	2,731
Small wading birds and shorebirds	*Glareola*	*maldivarum*	Oriental Pratincole	5,401	2,893
Small wading birds and shorebirds	*Stiltia*	*isabella*	Australian Pratincole	1,271	1,178
Small wading birds and shorebirds	*Himantopus*	*himantopus*	Black-winged Stilt	6,206	2,851
Small wading birds and shorebirds	*Recurvirostra*	*novaehollandiae*	Red-necked Avocet	373	373
Small wading birds and shorebirds	*Actitis*	*hypoleucos*	Common Sandpiper	2,174	1,829
Small wading birds and shorebirds	*Arenaria*	*interpres*	Ruddy Turnstone	472	472
Small wading birds and shorebirds	*Calidris*	*acuminata*	Sharp-tailed Sandpiper	147	147
Small wading birds and shorebirds	*Calidris*	*alba*	Sanderling	416	416
Small wading birds and shorebirds	*Calidris*	*canutus*	Red Knot	1,281	1,186
Small wading birds and shorebirds	*Calidris*	*ferruginea*	Curlew Sandpiper	678	667
Small wading birds and shorebirds	*Calidris*	*fuscicollis*	White-rumped Sandpiper	425	425
Small wading birds and shorebirds	*Calidris*	*ruficollis*	Red-necked Stint	3,271	2,412
Small wading birds and shorebirds	*Calidris*	*tenuirostris*	Great Knot	137	137
Small wading birds and shorebirds	*Gallinago*	*megala*	Swinhoe's Snipe	227	227
Small wading birds and shorebirds	*Limosa*	*lapponica*	Bar-tailed Godwit	4,661	2,824
Small wading birds and shorebirds	*Numenius*	*madagascariensis*	Eastern Curlew	260	260
Small wading birds and shorebirds	*Numenius*	*minutus*	Little Curlew	1,196	1,117
Small wading birds and shorebirds	*Numenius*	*phaeopus*	Whimbrel	2,851	2,216
Small wading birds and shorebirds	*Tringa*	*glareola*	Wood Sandpiper	959	917
Small wading birds and shorebirds	*Tringa*	*nebularia*	Common Greenshank	309	309
Small wading birds and shorebirds	*Tringa*	*stagnatilis*	Marsh Sandpiper	1,713	1,516
Small wading birds and shorebirds	*Xenus*	*cinereus*	Terek Sandpiper	2,729	2,152

Small wading birds and shorebirds	*Eulabeornis*	*castaneoventris*	Chestnut Rail	412	412
Small wading birds and shorebirds	*Gallirallus*	*philippensis*	Buff-banded Rail	850	821
Small wading birds and shorebirds	*Porzana*	*cinerea*	White-browed Crake	4,294	2,751
Small wading birds and shorebirds	*Porzana*	*pusilla*	Baillon's Crake	1,073	1,015
Small piscivores	*Chlidonias*	*hybridus*	Whiskered Tern	1,592	1,427
Small piscivores	*Chlidonias*	*leucopterus*	White-winged Black Tern	171	171
Small piscivores	*Larus*	*novaehollandiae*	Silver Gull	136	136
Small piscivores	*Sterna*	*albifrons*	Little Tern	2,150	1,814
Small piscivores	*Sterna*	*bengalensis*	Lesser Crested Tern	1,242	1,155
Small piscivores	*Sterna*	*bergii*	Crested Tern	611	605
Small piscivores	*Sterna*	*caspia*	Caspian Tern	1,530	1,380
Small piscivores	*Sterna*	*hirundo*	Common Tern	3,073	2,324
Small piscivores	*Sterna*	*nilotica*	Gull-billed Tern	10,663	1,066
Small piscivores	*Sterna*	*sumatrana*	Black-naped Tern	36,234	3,623
Large piscivores	*Pelecanus*	*conspicillatus*	Australian Pelican	10,351	1,035
Large piscivores	*Microcarbo*	*melanoleucos*	Little Pied Cormorant	14,906	1,491
Large piscivores	*Phalacrocorax*	*carbo*	Great Cormorant	8,204	2,216
Large piscivores	*Phalacrocorax*	*sulcirostris*	Little Black Cormorant	69	69
Large piscivores	*Phalacrocorax*	*varius*	Pied Cormorant	1,432	1,306

1.2 Conservation targets

To ensure our prioritization represented the whole distribution of rare species without over representing common ones, conservation targets for each species were chosen accordingly to the species' area of occupancy (Rodrigues *et al.* 2004). We set a fixed target corresponding to 100% of the range of species with an area of occupancy smaller than 500 km^2 (20% of species). We also set a fixed target of 10% of the range of species with an area of occupancy larger than 10,000 km^2 (24% of species). The target for species with area of occupancy of intermediate size (57% of species), was calculated using linear interpolation between these two limits (Fig S1).

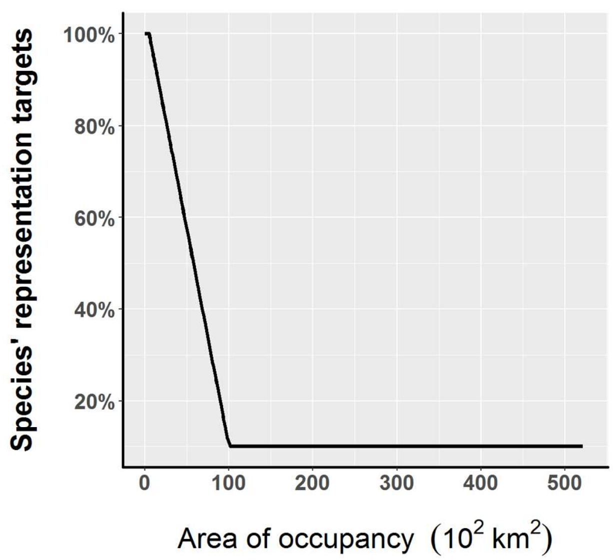

Figure S1. Relationship between area of occupancy and representation targets, for the 138 species in the Daly River catchment. For narrowly distributed species (area of occupancy < 500 km^2), targets equal 100% of their area of occupancy. For species which are more widespread (area of occupancy > 10,000 km^2). For species with area of occupancy of intermediate size, targets were calculated using linear interpolation.

1.3 Threat data

We focused on four major threats, for which data on the spatial distribution were available from Bartolo et al. (2008): (1) introduced water buffalos - *Bubalus bubalis*; (2) feral pigs - *Sus scrofa*; (3) grazing land use; and (4) para grass weed - *Brachiaria mutica*. These threats have a range of negative impacts on freshwater ecosystems (Pusey *et al.* 2011). Water buffalos and feral pigs increase riverbank erosion and the spread of weeds through trampling and rooting, thus increasing the amount of suspended sediment in the water column and reducing water clarity. Grazing (and other agricultural land uses) can increase sedimentation, nutrient enrichment, contamination with biocides and other chemicals, changes in run-off rates and increased likelihood of alien weed invasions. Aquatic weeds, such as para grass, form dense thickets on the water body surface, thus blocking sunlight from entering the water, reducing water oxygen levels and primary productivity, preventing water movement and reducing plant diversity and habitat availability for native fauna (waterbirds and small mammals).

We quantified the intensity of each threat in each planning unit by using categorical estimates of the relative incidence of buffalos, pigs, and para grass and the (continuous) aerial proportion of each planning unit (%) occupied with grazing land use (Bartolo, Bayliss & van Dam 2008; Australian Bureau of Agricultural and Resource Economics and Sciences 2010) (Fig. S2). We first scaled the raw intensity values of each threat, from 0 to 1, with 0 representing absence of the threat and 1 representing the highest intensity of the threat. We then classified the intensity of each threat into three uniform categories (Low, Medium and High).

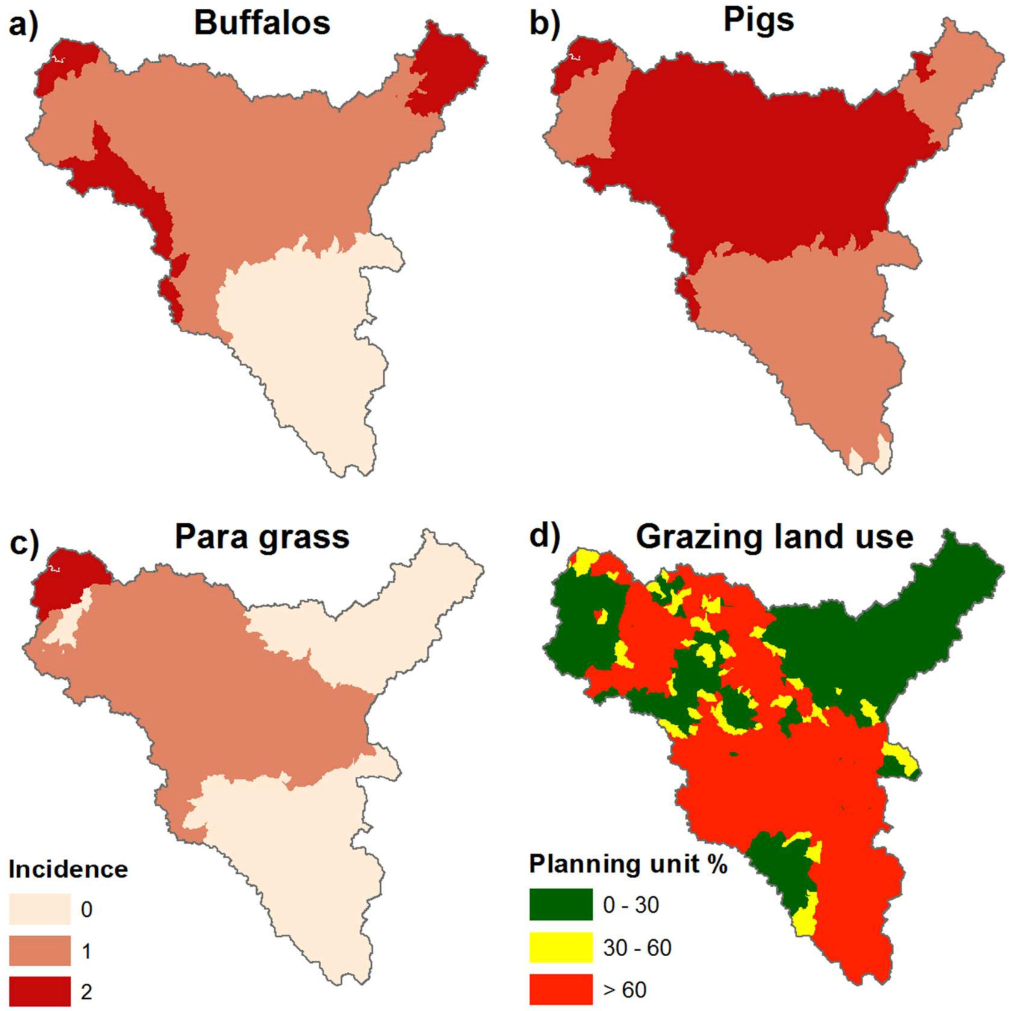

Figure S2. Spatial distribution of four major threats in the Daly River catchment, and their associated intensity (or magnitude). Incidence classes [for buffalos (a), pigs (b) and para grass (c)] are classified as follows: 0 = absent, 1 = occasional or localised occurrence, 2 = common and widespread, and 4 = abundant and widespread. For grazing land use (d), the proportion of each planning unit occupied by the land use is shown.

2. Expert elicitation

2.1 Overview

Expert elicitation is a robust and rigorous way to collect information for making conservation management decision, when empirical data is unavailable (Burgman *et al.* 2011; Martin *et al.* 2012). Although face-to-face interviews or workshop-based methods are more likely to elicit high-quality responses, remote means of elicitation (e.g., emails and online forum) are becoming popular in ecological applications (Donlan *et al.* 2010; McBride *et al.* 2012; Chades *et al.* 2015). This is due to the flexibility for experts of answering questions at their convenience (without having to be assembled in the same location at the same time) and the fact of having more time for digesting background materials and pondering answers. We adopted a structured, email-based elicitation approach following the five steps outlined by Martin et al. (2012): deciding how information will be used, asking the right question, designing the elicitation process, performing the elicitation and encoding the elicited information. The cast of actors in the elicitation process consisted in the decision analysts (the authors of this paper), analyst (LC) and experts.

2.2 Elicitation approach

We selected 5 experts for each faunal group of freshwater fishes, turtles and waterbirds. Experts were selected among Australian universities, State or Territory agencies and NGOs, for their extensive experience and expertise in the ecology and conservation of species in these faunal groups. Two of the authors (MJK and SL) initially contacted the experts to invite them to participate in an email-based survey and briefly explained project aims and outputs. Two of the experts who were initially contacted declined their availability and could not be replaced, due to time constraints.

The elicitation process was conducted following a modified Delphi method, whereby (1) the analyst (LC) and one of the co-authors (MJK) contacted the experts asking questions; (2) the experts provided their answer in a first round of elicitation; (3) the analyst summarized anonymously all the answers of the experts; (4) and the experts revised their answers, in a second elicitation round, providing a rationale to support them (McBride *et al.* 2012). Multiple rounds of elicitation maximize the chances of reaching a consensus among experts, which aids reducing the effect of between-expert variability. We developed an email-based survey consisting of (1) a set of background instruction with details about project background and aim, and instructions on how to complete the survey; and (2) an MS Office 2007 Excel spreadsheet containing the survey form. To expedite the expert elicitation process, we categorized the total 138 species (44 freshwater fishes, 8 turtles and 86 waterbirds) into 18 different ecological groups (6 ecological groups for each faunal group), based on body size and similarities in the use of habitat (movements) and food resources.

The survey was based on a 4-point elicitation procedure (Speirs-Bridge *et al.* 2010).We asked the experts to estimate: (1) the probability of persistence (ranging between 0 and 1) of species in each ecological groups, under the three intensities of each of the four threats; (2) a lower bound and (3) a upper bound, which represented the lowest and the highest plausible value, according to expert judgment, of probability of persistence; (4) and a level of confidence (ranging between 1 and 100 %) that the true value felt within the bounds. Lower and upper bounds around the species persistence estimates captured uncertainty in expert judgment regarding the true value of the responses. On the other hand, the level of confidence aids minimizing expert overconfidence during the elicitation process, i.e., minimizing the risk that the value provided by the experts lies outside of the lower-upper bound range.

Probability of persistence ranged between 0 and 1 and was defined as the likelihood that a species would exist over 20 years at high enough levels to perform its ecological function (Carwardine *et al.* 2012). The levels of persistence high enough to maintain ecological function for each species group were defined by the levels of habitat and resource use considered by the experts to be measureable and significant, as opposed to persistence levels which were no longer measurable or significant, equivalent to being ecologically extinct. The probability of persistence was estimated assuming that different threats did not interact. The whole approach was similar to the one followed by Carwardine et al. (2012) and Chades et al. (2015). Before round 2 of the elicitation, upper and lower bounds were normalized so that each expert had the same level of confidence (i.e. 80%) (McBride *et al.* 2012). This allowed experts' answers to be compared. Aggregated response values were produced by taking the arithmetic mean of best guesses, lower bounds and upper bounds across experts, for each ecological group, threat and threat intensity (McBride *et al.* 2012; Hemming *et al.* 2018). The group means represented the average best guess, and the average lowest and highest plausible response values, across the different experts. The elicitation process resulted in the collection of 72 ecological responses (18 ecological groups × 4 threats) (Fig. S3).

We prepared a different survey excel spreadsheet for each expert in each faunal group. Each spreadsheet included one table for each ecological group and threat combination, where the experts could insert their answers. We also set up self-updating line plots (one for each table), where expert answers were visualized as the values were typed in the tables. This allowed the experts to visualise the actual response "curve" (showing how persistence values change as threat intensity increases) and stimulate reasoning, thus facilitating the whole elicitation process and improving the precision and accuracy of the elicited information. The range of values that the experts could type was automatically restricted, to minimize the risk of inserting incorrect values. In the excel spreadsheet, we also included a worksheet with one worked example of how to correctly complete the survey; a list of the species in each ecological group; a table with a description of the intensity categories of different threats; and a summary table of the mechanisms of impact of different threat, sourced from Pusey et al. (2011).

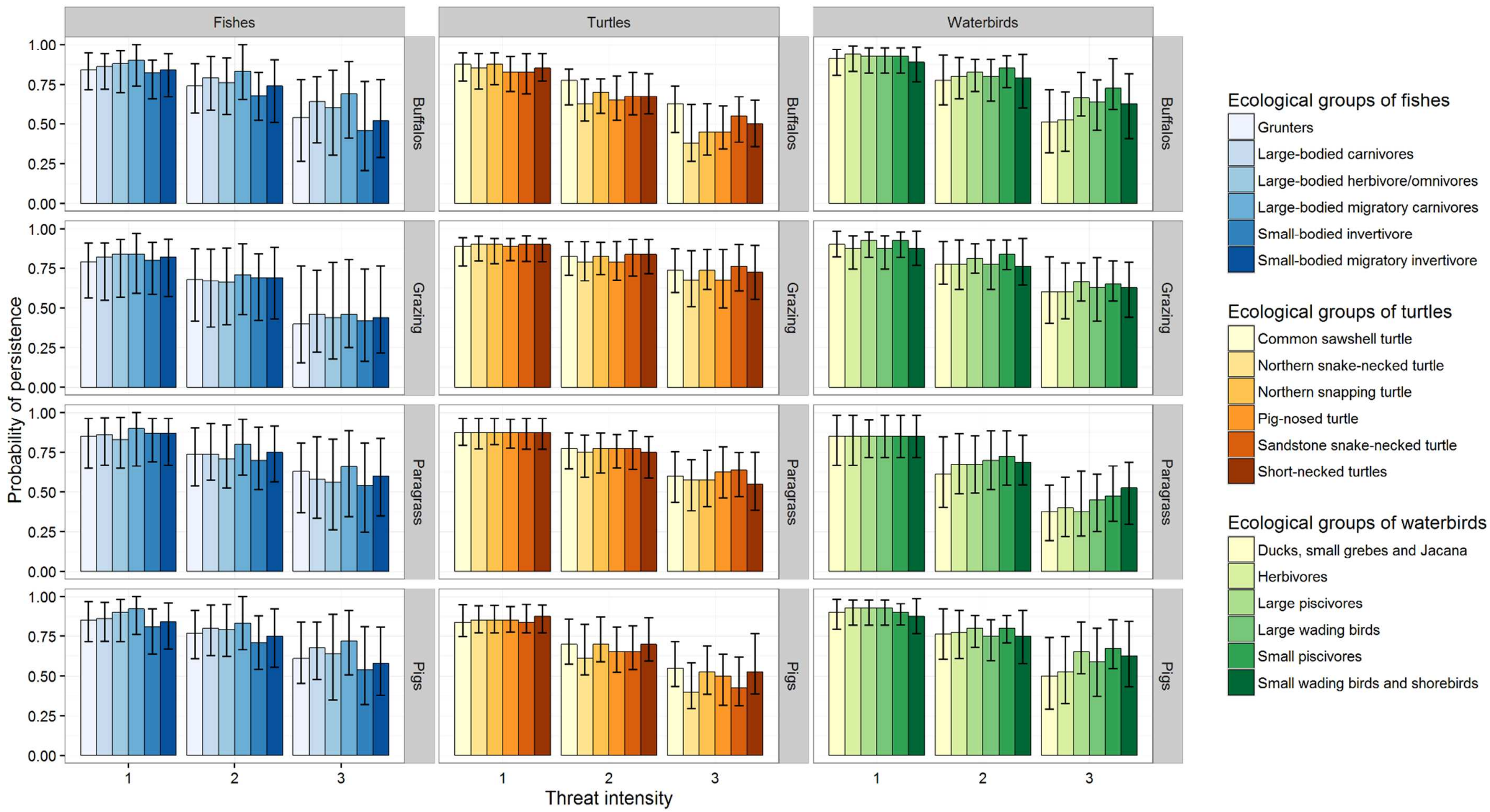

Figure S3. Species responses collected through expert elicitation. The values shown in the bar plots are the average (across experts) of the best guess estimates (bar height) and the average (across experts) of the upper and lower bounds (error bars).

3. Prioritization

3.1 Overview of prioritization objectives

Conservation prioritization problems differ in whether their objective is to maximize the biological value of species in areas where active management is carried out (e.g., protected areas) or to maximize the biological value of species across the entire landscape (including areas where no initial management is carried out) (Pressey, Watts & Barrett 2004; Polasky *et al.* 2005; Polasky *et al.* 2008; Moilanen, Possingham & Polasky 2009; Evans *et al.* 2015). The first objective assumes that the biological value of biodiversity features will decline in the absence of active conservation management, while the second assumes that species have a biological value even in the absence of prescribed management. The first objective tries to maintain representation of species which otherwise would go locally extinct, by selecting sites, actions and levels of management of effort (*representation* objective). The second objective tries to restore species biological value by selecting actions based on how much they improve the initial feature values (*restoration* objective). A common consequence of the representation objective is to focus on sites which are in good environmental conditions, because for similar conservation values, they are cheaper to act on. Conversely, the restoration objective will tend to focus on sites in bad conditions (high threat intensity), because gains (the improvement in conservation value relative to the initial conditions) are higher.

Planning for restoration is generally particularly sensible when threats are highly dynamics and are causing an ongoing reduction in the value of biodiversity features (Costello & Polasky 2004; Visconti *et al.* 2010). When threats are more uniform and static, planning for representation is more appropriate. The Daly river catchment is generally considered in good environmental conditions, with no major ongoing threats diminishing the value of biodiversity features (Schult & Townsend 2012; Adams *et al.* 2014). We therefore, decided to adopt a representation approach, which aims at representing a minimum target level for each species by selecting a set of levels of management effort and actions in sites which tend to be in good conditions.

3.2 Formulating the decision problem

Our aim was to identify which level of management effort to allocate to which action, and in which planning unit, to achieve representation targets for all species, while minimizing management costs. The *target* represents the probability of persistence of a species across each species' area of occupancy. It is expressed as the product of the probability of persistence, achieved through selected actions and effort, and the area of occupancy of a species in the planning units where actions and effort are selected.

We assumed that three potential levels of effort could be allocated to each action: (1) "Low", which keeps the threat at initial intensity; (2) "Medium", which reduces threat intensity by one category (e.g., from high to medium); and (3) "High", which reduces threat intensity by two categories (e.g., from high to low). The levels of effort available in each planning unit depended on the initial intensity of the threat, and they were therefore pre-specified for each planning unit. We assumed that threat intensity declines as a linear function of management effort (Wilson *et al.* 2011).

The decision problem is mathematically defined through the following objective function:

$$\min \sum_{i=1}^{N_p} \sum_{k=1}^{N_a} f_k\left(X_{i,k}\right) + Sp \qquad \text{eqn S3.1}$$

subject to achieving the representation target

$$\sum_{i=1}^{N_p} R_{i,j} \geq T_j, \qquad \text{for all species } j \qquad \text{eqn S3.2}$$

where $i \in \{1, 2, ..., N_p\}$, $k \in \{1, 2, ..., N_a\}$, $j \in \{1, 2, ..., N_s\}$ with N_p, N_a and N_s denoting the total number of planning units, actions and species, respectively. The first term in Equation S3.1 represents the sum of the costs of selected levels of management effort, where $X_{i,k}$ is a control matrix indicating the level of effort selected for action k in planning unit i, with $X_{i,k} \in \{1, 2, 3\}$; and f_k $(X_{i,k})$ is the cost (net present value in Australian dollars) of implementing level of effort $X_{i,k}$ for action k in planning unit i (see section below on how costs were calculated). The second term in equation S3.1 represents the species penalty, which is a measure of how the representation of each species in the solution is far from the target (see description below and equation S3.4). In equation S3.2, $R_{i,j}$ is the representation level of species j in planning unit i achieved through the selected levels of effort, and T_j is the target level for species j.

We assumed that different actions had an additive impact on species representation in a planning unit (i.e., no interaction between the impacts of different actions) (Auerbach *et al.* 2015). The representation level, $R_{i,j}$, of species j in planning unit i was expressed as follows:

$$R_{i,j} = a_{i,j} \times \frac{\sum_{k=1}^{N_a} d_{i,k}[g_{k,j}(X_{i,k})]}{Z_{i,j}} \qquad \text{eqn S3.3}$$

where $a_{i,j}$ is the area of occupancy of species j in planning unit i, $g_{k,j}$ $(X_{i,k})$ is the probability of persistence of species j following implementation of effort $X_{i,k}$ for action k in planning unit i (see section below on benefits calculation), $d_{i,k} \in \{0,1\}$ is a control variable indicating weather action k is available in planning unit i, and $Z_{i,j} = \sum_{k=1}^{N_a} d_{i,k}[g_{k,j}(X_{\max})]$ is the sum of the probabilities of persistence of species j, achieved by selecting the highest level of effort available for all actions in planning unit i (X_{max}). Equation S3.3 scales the area of occupancy of a species in a planning unit by the probability of persistence of the species following implementation of selected available effort. By doing so, we interpreted the representation level of a species in a planning unit as a "persistence area", or, in other words, as a measure of the probability of persistence of a species across its area of occupancy. Dividing by $Z_{i,j}$ ensures that the representation level of a species in a planning unit is proportional to the effort required to completely eradicate all threats to the species. For instance, if there are two actions available in a planning unit, and only one action is selected with a level of effort of 3 (highest), assuming $g_{k,j}$ is a linear function for both actions and the species occupies 10 km^2 of the planning unit (i.e., $a_{i,j}$= 10), $R_{i,j} = 10 \times \frac{1}{2} = 5$.

To ensure the achievement of targets we calculated a species penalty which was a function of the amount of target that had not been met, for each species. The cumulative species penalty, Sp, for all species N_s was calculated as follows:

$$Sp = \sum_{j=1}^{N_S} SPF_j H(s_j) s_j \qquad \text{eqn S3.4}$$

where SPF (Species Penalty Factor) is a scaling factor which determines the relative importance of meeting the target for each species. The Species Penalty Factor was set to 10, which was the minimum value to ensure all targets were 100% met. The step function, $H(s_j)$, takes a value of zero when $s_j \leq 0$ and 1 otherwise. The shortfall s_j represents how much of the representation target for each species is not met and is equal to $T_j - \sum_{i=1}^{Np} R_{i,j}$. Calculating the shortfall jointly over all planning units (N_p) ensures that the set of priority planning units and actions collectively provides the greatest contribution in terms of achieving conservation goals (i.e., principle of complementarity) (Moilanen, Possingham & Polasky 2009).

3.3 Benefits of management effort

The benefits of implementing a particular level of effort of an action was expressed in terms of improved probability of persistence of the species. The probability of species persistence $g_{k,j}$ ($X_{i,k}$), following selection of effort $X_{i,k}$ for action k in site i, depended on the species responses to the threats, which were derived through expert elicitation. The specific value of $g_{k,j}$ ($X_{i,k}$) was equal to the probability of persistence under the intensity of the threat achieved following implementation of effort $X_{i,k}$ (Fig. S4). For instance, given an initial "High" threat intensity, the benefit of implementing a "Medium" level of effort for an action ($X_{i,k}$ = 1) corresponded to the species persistence under "Medium" intensity of that threat (i.e., "P2" in Fig S4). The benefit of the "Low" effort ($X_{i,k}$ = 1) corresponded to the probability of persistence under the initial intensity of the threat.

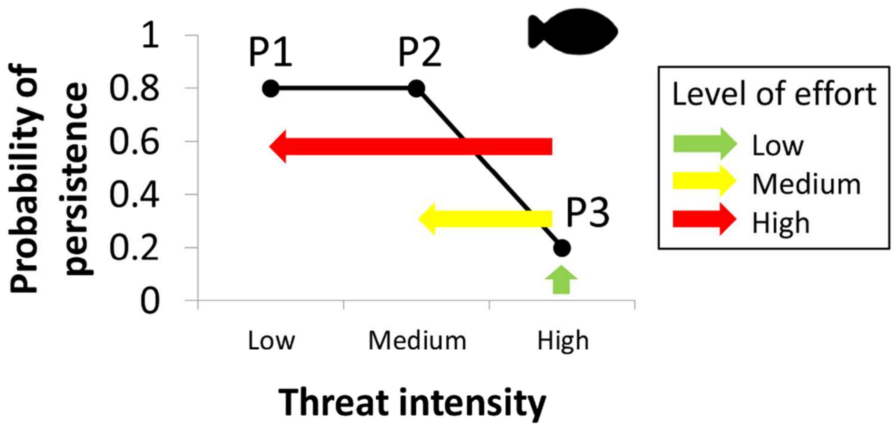

Figure S4. Schematic diagram describing how the benefits of different levels of effort (Low, Medium and High), for each action, were calculated from the species responses to the threats. "Low" keeps the threat at initial intensity; "Medium" reduces threat intensity by one category (e.g., from high to medium); and "High" reduces threat intensity by two categories (e.g., from high to low). The benefit of a given level of effort was equal to the species probability of persistence under the intensity of the threat achieved following implementation of the given effort.

3.4 Costs of management effort

We calculated the management costs of implementing different levels of effort for each action (aerial shooting of water buffalo, aerial shooting of feral pigs, building cattle fences and chemical spraying of para grass) in the Daly River catchment. We sourced cost of long-term capital, materials, supplies and labour for different actions from reference studies conducted in tropical northern Australia, where threatening processes operates within contexts (i.e., climate, vegetation communities) similar to the Daly River catchment.

The costs reported in the reference studies were not directly comparable against each other, as they were calculated at disparate spatial scales and management time horizons. We therefore standardized reported costs by calculating costs per hectare of each action. To standardize cost estimates across studies, we: (1) calculated the total annual cost of implementing each action over the entire region of the reference study; (2) corrected the figures obtained in step 1 to account for the variations in the costs due to implementation of the action over a 20-year time frame (e.g., chemical spraying costs decline over time as the size of infestation is reduced); (3) calculated costs per hectare by dividing annual regional costs by the area that was treated with the action ("treated area of the region"), within the entire region of the reference study; (4) used costs per hectare derived from the reference studies to estimate the cost of implementing different levels of effort of each action in each of the planning units of the Daly River catchment.

Firstly, we extrapolated from the reference studies the annual management costs of implementing each action, over the entire region, over 20 years. We then estimated the total cost now (Net Present Value) of implementing each action, over 20 years, using a 5% discount rate (Carwardine *et al.* 2012). To calculate the Net Present Value (*NPV*), we measured the present value of a series of equal payments over a number of time series, using the following present value equation (Carwardine *et al.* 2011):

$$NPV = \frac{C_{annual} \times t}{(1+r)^t} \qquad \text{eqn S3.5}$$

where C_{annual} is the annual regional management cost, t is the time frame, which in our case was 20 years, and r is the discount rate, which in our case was 5%.

Secondly, we calculated, for each action, the Average Equivalent Value (*AEV*), which represents the annual cost of implementing the action over the entire region, corrected for the effect of the discount due to investment in the action over multiple years (Carwardine *et al.* 2011). The *AEV* was calculated as it follows:

$$AEV = NPV \times \frac{r}{1 - \frac{1}{(1+r)^{t+1}}} \qquad \text{eqn S3.6}$$

Thirdly, we divided the *AEV* by the area of the entire region of the reference study, to estimate the management cost (AU$/ha/year) of implementing each action, in one hectare of the region, over one year (Table S2). When estimating costs per hectare, we accounted for the area where the action was carried out ("treated area of the region"), within the entire region, as described in the reference study. For example, the Average Equivalent Value of conducting chemical weed spraying in a 2,000,000ha region is AU$ 200K per year. However, if chemical spraying is only carried out in 1/3 of the region (treated area of the region ~ 600,000 ha), the cost per ha, per year, is around AU$ 0.33 (200,000/600,000). Details of how we estimated the treated area of the region for each of the four actions are reported in the sections on individual actions below and in Table S2.

Finally, we assumed that the cost of implementing different levels of effort of each action in each of the planning units of the Daly river catchment was a linear function of the cost of implementing the action over the whole treated area of a planning unit. The cost $f_k(X_{i,k})$ of implementing level of effort $X_{i,k}$ for action k in planning unit i was calculated as follows:

$$f_k(X_{i,k}) = \frac{X_{i,k}}{NE_{i,k}} \vartheta_k \; TArea_i \qquad \text{eqn S3.7}$$

where $X_{i,k} \in \{1, 2, 3\}$, is the selected level of effort, with 1 = "Low" level of effort, 2 = "Medium" level of effort and 3 = "High" level of effort; $NE_{i,k}$ is the total number of alternative level efforts available for implementation of action k in planning unit i; ϑ_k is the cost per hectare of implementing action k and $TArea_i$ is the area of the planning unit i where the action was applied ("treated area of the planning unit"). The coefficient $\frac{X_{i,k}}{NE_{i,k}}$ ensures that the costs of different levels of effort are expressed as proportions of the cost of implementing an action in the whole treated area of a planning unit. For instance, the cost of implementing the "Low" level of effort $(X_{i,k} = 1)$, in a planning unit where there are two levels of management effort available for implementation for that action $(NE_{i,k} = 2)$, is half the cost of implementing the action in the whole treated area of the planning unit $(\frac{1}{2} \times \vartheta_k TArea_i)$. We assume that the cost of implementing the highest level of effort corresponded to the cost of implementing the action in the whole planning unit. Details on how we calculated the treated area of the planning unit for each action are described in the sections on individual actions below.

When estimating management costs, we made three main assumptions: (1) the cost of implementing an action in the Daly River catchment was the same as the cost of implementing the action in the region from where the cost estimate (AU$/ha/year) was derived; (2) the cost of an action was homogenous across different parts of the Daly River catchment. In other words, we assumed that costs did not vary across the study area due, for example, to differences in topography or land value (which might increase the cost of implementing some actions in some part of the catchment, where more effort is required or a certain level of effort is simply is more expensive). Finally, (3) we assumed that AU$/ha/year for each action were constant as the area over the effort is allocated increased. In other words, we ignored "economy of scale", which causes costs per unit of area to decrease as the area (which is a measure of "effort"), over which the action is applied, increases (Adams, Pressey & Stoeckl 2012).

Below, we describe in more detail: (1) which activities are considered in the management costs derived from the reference studies; (2) how we derived annual regional costs from the reference studies; (3) what was the regional area and the treated area of the region in the reference study; and (4) how we calculated the treated area of each planning unit for each action.

3.4.1 Aerial shooting (buffalos and pigs)

We used cost estimates of implementing aerial shooting of water buffalos and feral pigs within the Kakadu National Park (KNP) in northern Australia (Bayliss & Yeomans 1989; McMahon *et al.* 2010). We estimated the cost of aerial shooting from Bell-47 helicopters, using the STAR model developed in McMahon et al. (2010). STAR is a spatially-explicit model for estimating the management costs of controlling (i.e., culling) large feral ungulates (pigs, water buffalo and horses) within the KNP region. We considered the costs of decreasing population size down to 75% of its original value (in the first year) and the annual cost of culling for maintaining the population at the 75% level (every following year). We used the STAR model to calculate the total annual cost of aerial shooting, for water buffalos and feral pigs separately, over the entire KNP region. We assumed that aerial shooting for buffalos took place mainly on the floodplain (treated area of the region = 1/3 of the total KNP extent ~ 600,000 ha), where the largest buffalo population is located, while feral pig shooting took place across approximately half of the entire KNP region (treated area of the region = 990,000 ha). Thus, when calculating the costs of management per hectare, we divided the Average Equivalent Value by 600,000 ha, in the case of buffaloes, and by 990,000 ha, in the case of pigs (Table S2).

We assumed that the area of each of the planning units in the Daly River catchment that had to be treated with aerial shooting of buffalos and pigs ("treated area of the planning unit) equalled the area of the entire planning unit. Invasive herbivores are highly mobile and often capable of moving large distances, which often requires covering large distances in helicopter (Bayliss & Yeomans 1989; McMahon *et al.* 2010).

3.4.2 Riparian fencing

We used cost estimates of fencing riparian vegetation to reduce grazing pressure on riverine habitat, on individual farming properties, in the Kimberley region of Northern Australia (Carwardine *et al.* 2012). We extracted, from the study, the reported total annual cost of materials and labour for building, and maintaining over 20 years, cattle fence in one individual property of the region. Because Kimberley properties are large (average property size in the Kimberley is around 200,000 ha), we treat the property as the entire region. However, we assumed that riparian fencing would be carried out across 1% of each property, resulting in a total fenced area per property of 2,000 (treated area of the region) (Table S2). Thus, when calculating the costs of management per hectare, we divided the Average Equivalent Value by 2,000 ha.

Riparian fencing occurs along the river banks where riparian vegetation is located. Fencing costs depend on the length of the fence that needs to be built (Clapperton & Day 2001), and therefore depend on the length of the river. However, our unitary costs are derived using regional costs, at the individual property scale, and then averaging across the area of the property occupied by the river (1%). This means that the resulting cost per hectare ignores the length of the river that was actually fenced within the property. It might be that some hectares (within the property) had 400 m of fence (all 4 sides), some had 200 m of fence (2 sides), and some others had 100m of fence (1 side). To be able to calculate costs of fencing as a function of river length using our unitary costs, we assumed that the treated area of the planning unit for fencing corresponded to the area of each of the planning units of the Daly River catchment occupied by the river. We calculated the river area in each planning unit, by assuming an average river length of 100 meters.

3.4.3 Chemical spraying

We used cost estimates of chemical spray of para grass infestations in Kakadu National Park, conducted with the goal of local eradication, using burning and spraying with herbicide (Glyphosate 450®) (McMaster *et al.* 2014). We considered total costs of control on an annual basis (labour, vehicle and chemical) and assumed that the annual costs of control decrease every year as the density of the infestation is reduced, using a decision rule of thumb developed by Adams and Setterfield (2013). The rule of thumb assumes a maximum control period of 10 years and an ongoing annual monitoring cost after the control period is completed. McMaster et al (2014) reported costs at a management unit scale of 6.25 ha, which was much smaller than the management unit scale reported for the other actions, i.e., the entire region, or the property). Therefore, we extrapolated these costs to the regional scale assuming that control was carried out over all mapped infestations, *c.a.* 12,500 ha of the total Kakadu area (Table S2). When calculating the costs of management per hectare, we divided the Average Equivalent Value by 12,500 ha.

We assumed that the treated area of the planning unit for chemical spraying corresponded to the area of para grass infestation in the planning unit. Para grass is an aquatic weed, which only occur in river bodies, thus control does not require searching the entire planning unit (Bartolo, Bayliss & van Dam 2008). We derived estimates of para grass infestation area in each planning units of the Daly by converting relative incidence classes to proportions of para grass cover in the planning units, using conversion tables for para grass infestation in other regions of northern Australia (McMaster *et al.* 2014). We assumed that incidence classes 0, 1, 2 and 4 corresponded to 0, <10%, 10-50%, >50% proportion of para grass cover in the planning unit, respectively. To calculate the area of para grass infestation in each planning unit, we multiplied the area of the planning unit by the average of the range of proportions of para grass cover for a specific incidence class. So, if the incidence of para grass in a planning unit was 2 (corresponding to 10-50%) and the planning unit had an area of 10 km^2, the area of para grass infestation (and therefore also the “treated area of the planning unit”) equalled 3 km^2 (0.3 × 10).

Table S2. Description, treated area and cost per hectare for the four different management actions sourced for reference studies in northern Australia. “Treated area” refers to the area of the entire region of the reference study where the action was carried out.

Action	Description	Treated area (ha)	Cost per hectare (AU$/ha/year)	Reference
Aerial shooting to control buffalo	Shooting from Bell-47 helicopters	600,000	0.62	Bayliss and Yeomans 1989; McMahon et al. 2010
Aerial shooting to control pigs	Shooting from Bell-47 helicopters	990,000	1.63	Bayliss and Yeomans 1989; McMahon et al. 2010
Fencing riparian zones to prevent stock access	Installing cattle-proof fence	2,000	35.28	Carwardine et al. 2011
Burning and chemical spraying to control para grass	Burning and spraying herbicide (Glyphosate 450®) over para grass infestation	12,500	47.42	McMaster et al. 2014

3.5 Solution method

We adopted an approach similar to the one underpinning reserve selection software, such as Marxan (Ball, Possingham & Watts 2009). The algorithm developed here, as the Marxan one, uses simulated annealing to find a near-optimal solution (Kirkpatrick, Gelatt & Vecchi 1983). Simulated annealing is a mathematical optimization technique which uses a stochastic search designed to escape local optima when searching for a global optimal solution. Essentially, the simulated annealing algorithm works by iteratively inducing a random change in the status of a system and then evaluating an objective function. Changes that reduce the values of the objective function are always accepted, while changes that do not reduce, or increase, the value of the objective function are accepted with a probability which decreases as the annealing proceeds. This technique, similar to the cooling of metals, aids the algorithm to escape non-optimal solutions. The simulated annealing algorithm was implemented in the R programming language for statistical computing (R Development Core Team 2013).

Our simulated annealing algorithm iteratively removes from, or adds to, the solution, one level of management effort for one action in one planning unit, at the time. The control variable $\mathbf{X} = [X_{ik}]$ was modelled as a planning units × actions matrix, which indicates the level of effort allocated to each action k in each planning unit i. In order to find a near-optimal combination of levels of effort, actions and planning units, the algorithm minimizes the value of the objective function by iteratively changing the value of the control variable. For each optimization routine, the algorithm generates an initial value X of the control matrix by allocating one level of effort, $X_{i,k} \in \{1, 2, 3\}$, to an initial proportion, P_i, of all actions available in all planning units. An action is available in a planning unit if the threat that the action abates is present in the planning unit. The initial value of the objective function is then calculated.

At each of the following iterations, a modified value X' of the control matrix is generated by drawing one planning units × actions × efforts combination at random from a uniform distribution of all planning units × actions × efforts combinations. The selected level of effort is then allocated to the selected action in the selected planning unit. The selected level of effort substitutes the level of effort already in the solution for that planning unit × action combination. The new value of the objective function after the change is calculated and negative changes are always accepted. Positive changes in the value of the objective function are accepted with probability:

$$P = e^{\frac{-\Delta_{OF}}{Temp}}, \qquad \text{eqn S3.8}$$

where

$$Temp = Temp_0 \times \alpha , \qquad \text{eqn S3.9}$$

with Δ_{OF} being the difference in the value of the objective function after change X', $Temp$ the current temperature in the cooling schedule, $Temp_0$ the initial temperature value, and α the cooling factor. At each iteration, the initial temperature value decreases according to equation S3.9. Therefore, as the cooling schedule progresses, P decreases, and only positive changes in the value of the objective function are accepted. We employed an initial temperature of 1 and a cooling factor of 0.99999.

4. Effect of expert knowledge uncertainty on target achievement

4.1 Overview

Expert knowledge is widely used to aid making conservation management decisions (Burgman 2005; Sutherland 2006; Kuhnert, Martin & Griffiths 2010). Given the lack of empirical estimate about the benefits of management action for species, expert knowledge represents an invaluable resource to estimate the responses of species to management actions, in prioritization studies (Burgman, Lindenmayer & Elith 2005; Runge, Converse & Lyons 2011; Martin *et al.* 2012). However, similarly to empirical knowledge, information derived through expert knowledge have an associated uncertainty. Uncertainty in expert knowledge can stem from a variety of sources, including imperfect knowledge of the system, natural variability in the ecological process and language uncertainty when communicating with experts (Regan, Colyvan & Burgman 2002). Management decisions that do not account for uncertainty can lead to suboptimal decisions, which might increase species risk of extinctions, for example (McDonald-Madden *et al.* 2010).

The most important uncertainties to focus on is those that, when reduced, can improve the outcome of the management decisions (Runting, Wilson & Rhodes 2013; Maxwell *et al.* 2015). Natural variability might not be significantly reduced as it inherent in every system and language variability might be easily reduced by improving elicitation methods. Epistemic uncertainty, which derives from imperfect knowledge of a system, might be reduced by acquiring more knowledge (Grantham *et al.* 2009). Therefore, understanding the impact that epistemic uncertainty has on the prioritization outcome might shed light on whether it is worth investing in acquiring additional information (on the responses of species to actions, for instance) or make a decision with the available information.

The main sources of epistemic uncertainty which might bias the information acquired during an expert elicitation process are mainly two: (1) variability in the answer provided by individual experts, and (2) variability in the answer provided by different experts (Runge, Converse & Lyons 2011; McBride, Fidler & Burgman 2012). The variability in the answer provided by each individual expert is commonly represented by a as a range of values bounded between a lower and an upper estimate (*lower bound* and *upper bound*), around an expert most likely answer (*best guess*) (Speirs-Bridge *et al.* 2010). We refer to this type of uncertainty as an expert's own confidence level. The variability in the answer provided by different experts (who might have different levels of expertise, experience, judgement) is represented by the range of values provided by different experts, to which we refer as across-expert variability. Here we focus on the variability in each expert's own answer (*lower bound* and *upper bound*).

4.2 Approach

We quantified the impact of an expert's own confidence level on the achievement of conservation objectives by using a two-stage approach. We first used a solution generated through the spatial prioritization approach where best guesses (averaged across experts) were assumed to be the 'true' responses of species to threats (and thus used in the prioritization). These spatial priorities reflect real-world cases in which managers use best available information to set priority locations and then use these maps to direct on-ground effort. We called this analysis 'best guess scenario'. We then estimated the extent to which implementing the best guess scenario priorities may result in either over or under-achievement of species targets, when the true species responses deviate from expert best guesses due to an expert's own confidence level.

We calculated the representation of each species in a prioritization solution assuming that the true response corresponded to either the lower bound or the upper bound of the experts' answers (averaged across individual experts), respectively. The responses of each taxa in different faunal groups were averaged across the experts in each faunal group, by taking the mean of the experts' best estimates, and the mean of the upper and lower bounds (after normalising individual estimates to fit 100 % credible bounds).

To measure the degree to which uncertainty in expert knowledge affected achievement of conservation targets, we calculated the percentage change (averaged across species) between representation of a species when true responses were assumed to be different from the averaged best guesses, and representation of a species in the best guess scenario (Figure 3 in the main text). To better characterize under or overachievement of targets, we also calculated for each scenario: (1) the proportion of the total number of species represented in the prioritization solution below and above target level, and (2) the percentage change in the representation of each species relative to their targets (Figure S5-S6).

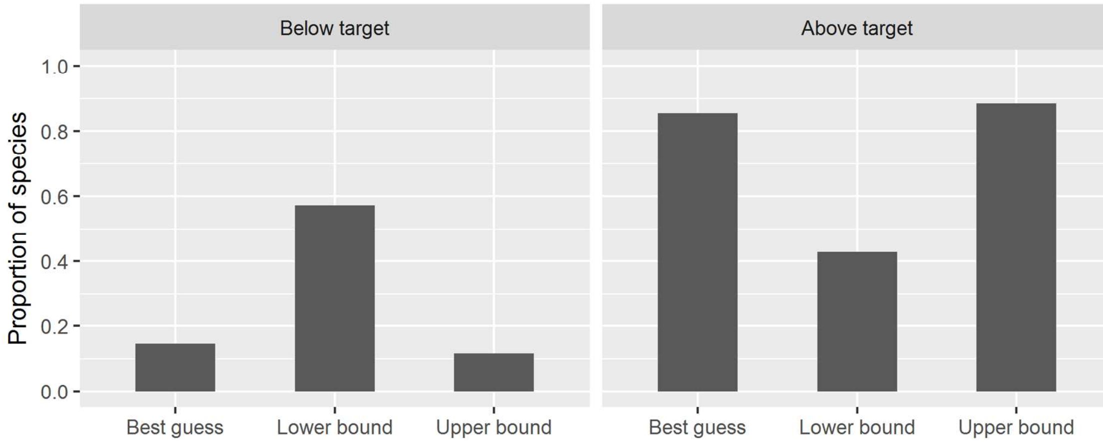

Figure S5. Proportion of species represented below and above target level in the prioritization, when true species responses were assumed to be known without uncertainty ("Best guess") and when true species responses were assumed to be lower and greater than the experts best guess ("Lower bound" and "Upper bound"), due to uncertainty in expert knowledge.

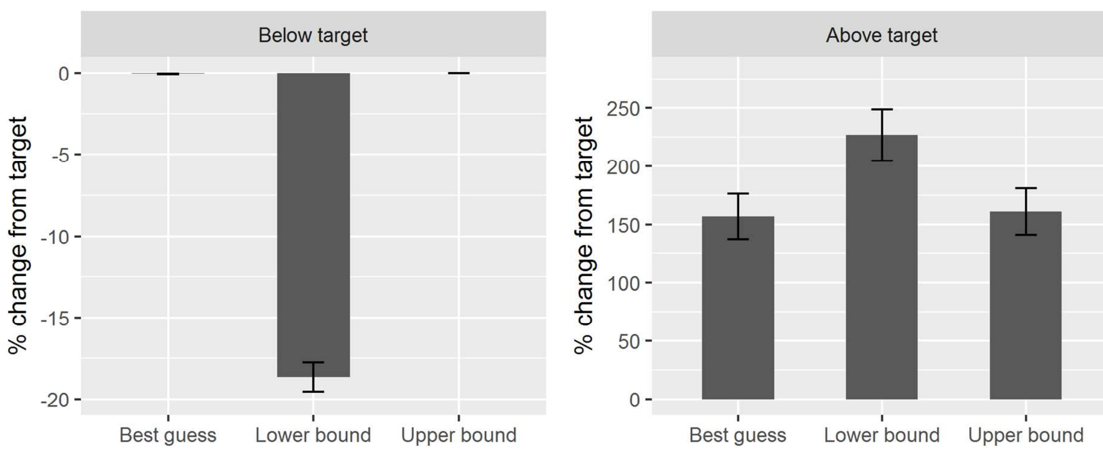

Figure S6. Percentage change (mean ± 1 SE) in the representation of each species relative to their targets, when true species responses were assumed to be known without uncertainty ("Best guess") and when true species responses were assumed to be lower and greater than the experts best guess ("Lower bound" and "Upper bound"), due to uncertainty in expert knowledge. Percentage changes below and above targets were averaged across species separately.

4.3 Effect of varying target level

We quantified how varying target level influenced the impact of expert knowledge's uncertainty on species representation error. The effect of the uncertainty's lower bound was almost constant across a range of different target levels, with average species representation being 21-20% lower relative to the best guess scenario, across all target levels (Figure S7). As target levels increased the proportion of species which did not meet their conservation targets increased (Figure S8). These species achieved approximately 20% of their conservation targets; a pattern which was constant across target levels (Figure S9).
The effect of the uncertainty's upper bound on species representation error was generally smaller compared to the effect of the uncertainty' lower bound. This was due to the fact that most of the species were already represented above target levels in the prioritization (Figure S8). Therefore, assuming that the experts' upper bounds were the true species responses did not make a large difference relative to the best guess scenario, because species were already above targets.

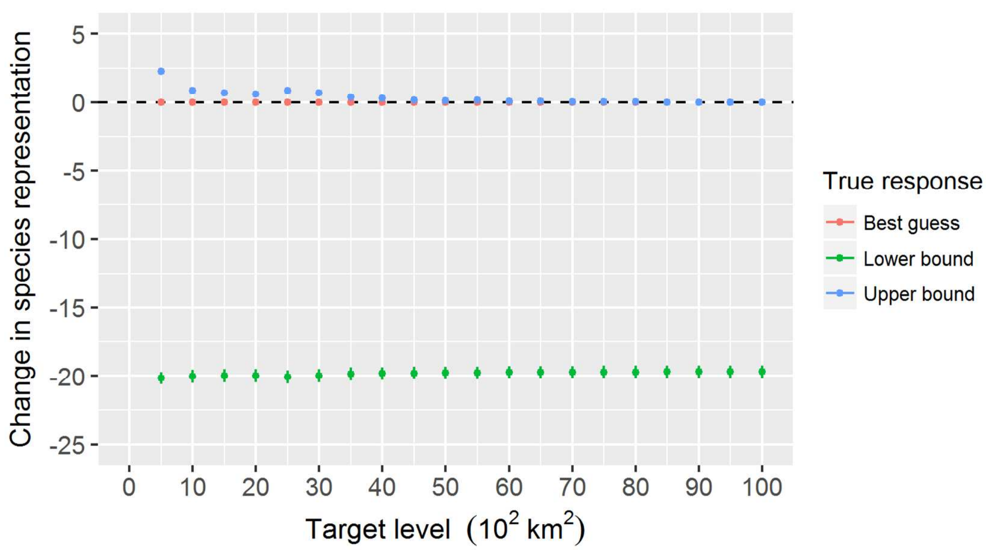

Figure S7. Percentage change in species representation, relative to the best guess scenario, when the true species responses deviate from an expert best guess, due to an expert's own confidence level, for different target levels. The value on the y axis is the percentage change value averaged across species (± 1 SE). Best guess scenario refers to when the best guesses of individual experts (averaged across experts) were assumed to be the true species responses (dashed line). The effect of an expert's own confidence level refers to when the lower and upper bounds of individual experts (averaged across experts) were assumed to be the true species responses. Displayed values are from the run with the lowest objective function, among a set of 10 replicate runs.

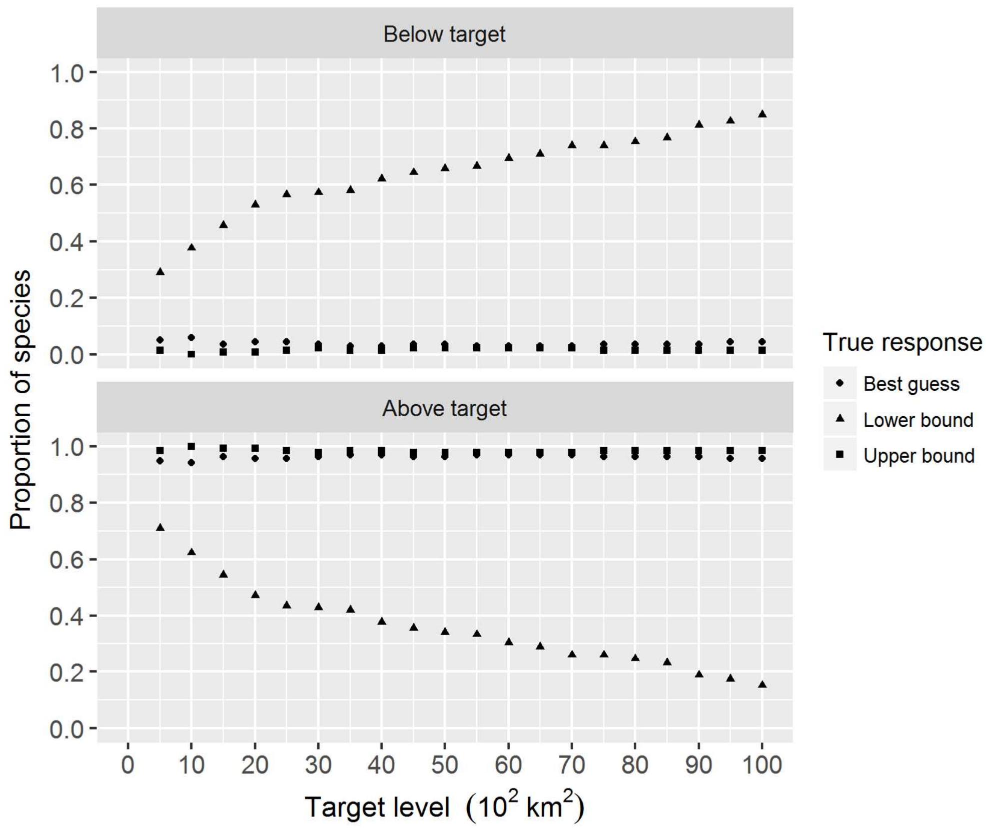

Figure S8. Proportion of species represented below and above target level in the prioritization, when the true species responses deviate from an expert best guess, due to an expert's own confidence level, for different target levels.

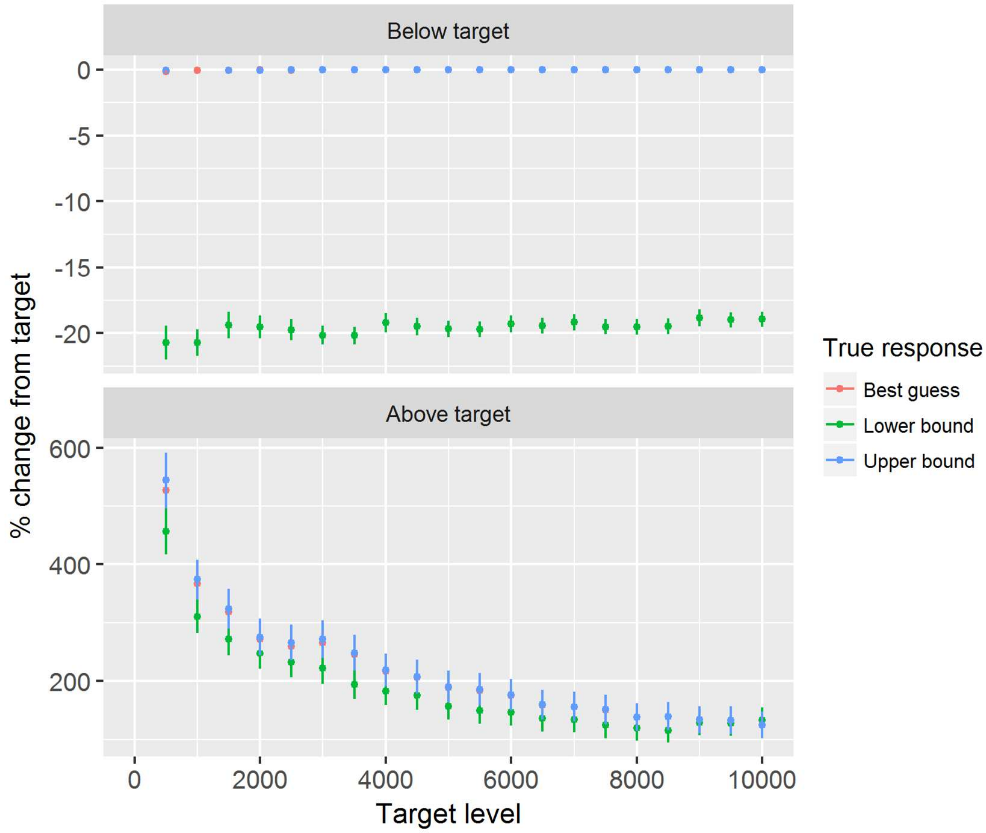

Figure S9. Percentage change (mean ± 1 SE) in the representation of each species relative to their targets, when the true species responses deviate from an expert best guess, due to an expert's own confidence level, for different target levels. Percentage changes below and above targets were averaged across species separately.

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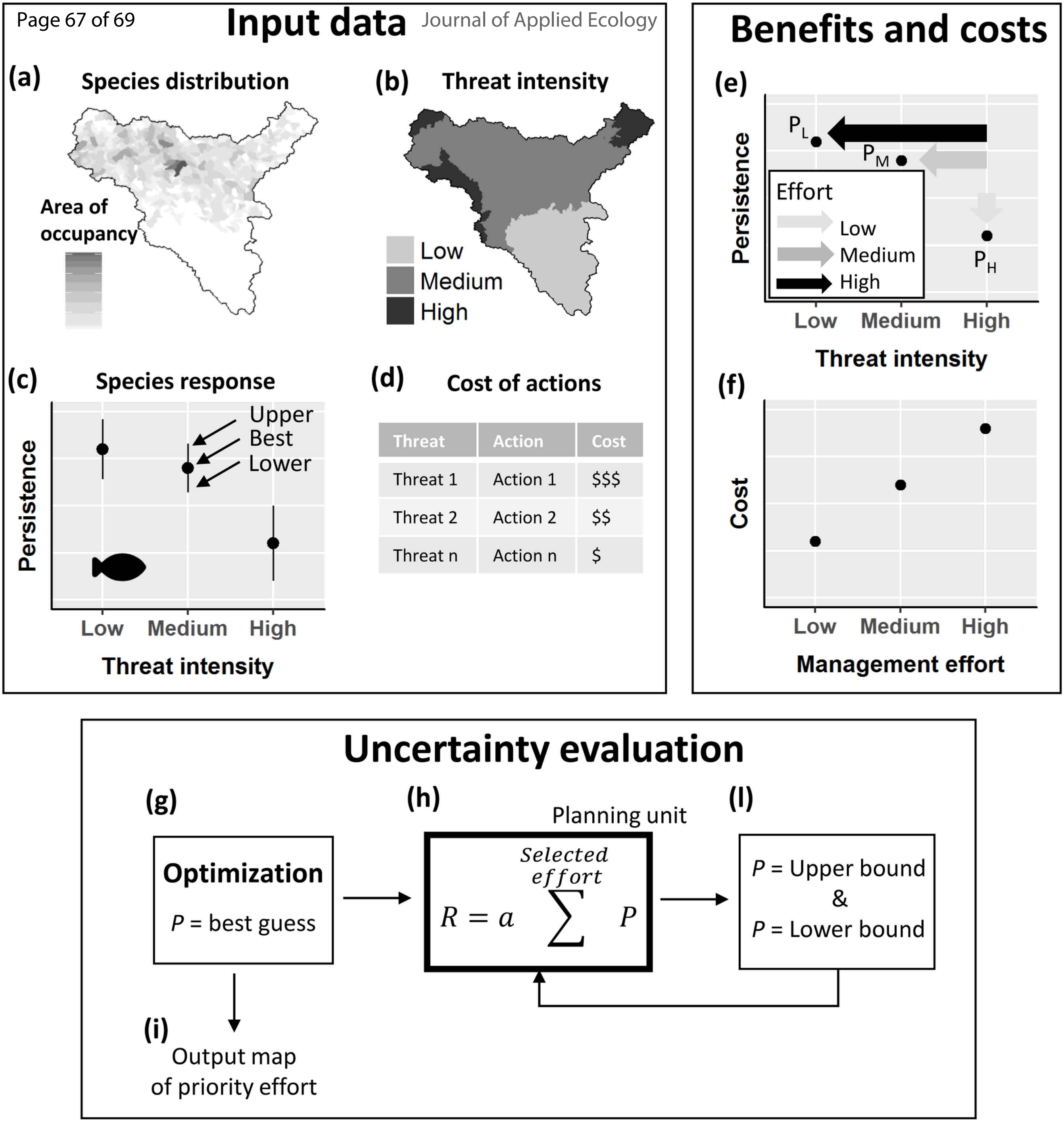
Input data
(a) Species distribution
Area of occupancy
(b) Threat intensity
Low
Medium
High
(c) Species response
Upper
Best
Lower
Persistence
Low Medium High
Threat intensity
(d) Cost of actions
Threat | Action | Cost
Threat 1 | Action 1 | $$$
Threat 2 | Action 2 | $$
Threat n | Action n | $
Benefits and costs
(e)
P L
P M
P H
Effort
Low
Medium
High
Persistence
Low Medium High
Threat intensity
(f)
Cost
Low Medium High
Management effort
Uncertainty evaluation
(g)
Optimization
P = best guess
(h)
Planning unit
R = a Σ P
Selected effort
(l)
P = Upper bound
&
P = Lower bound
(i)
Output map
of priority effort

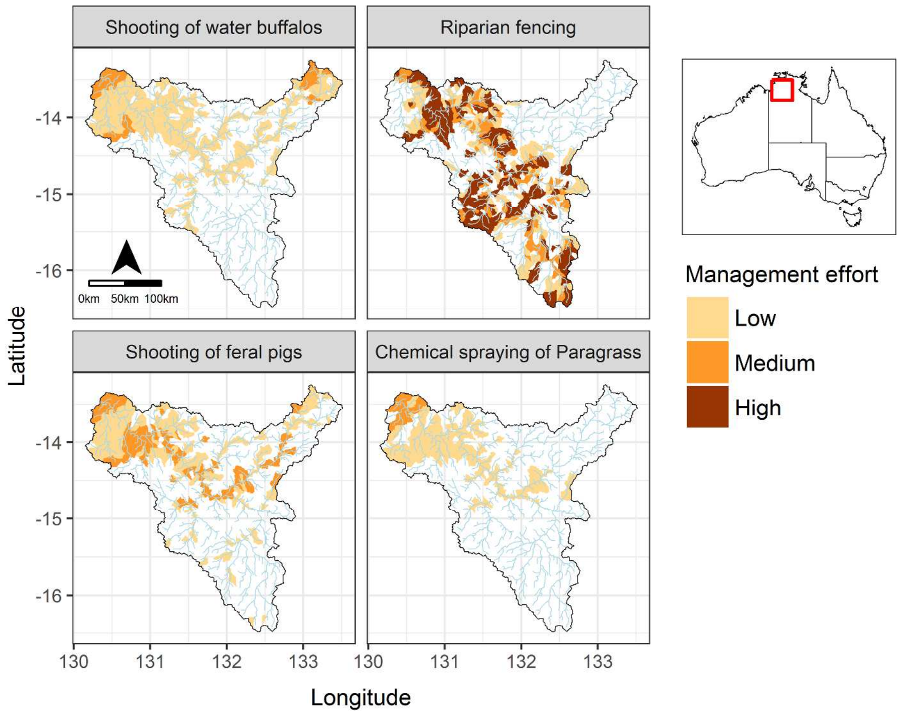

Figure 2. Spatial distribution of management effort allocated to four different actions in the Daly River catchment. Results are shown for the best solution of 10 replicates and best guess expert estimate (averaged across experts).

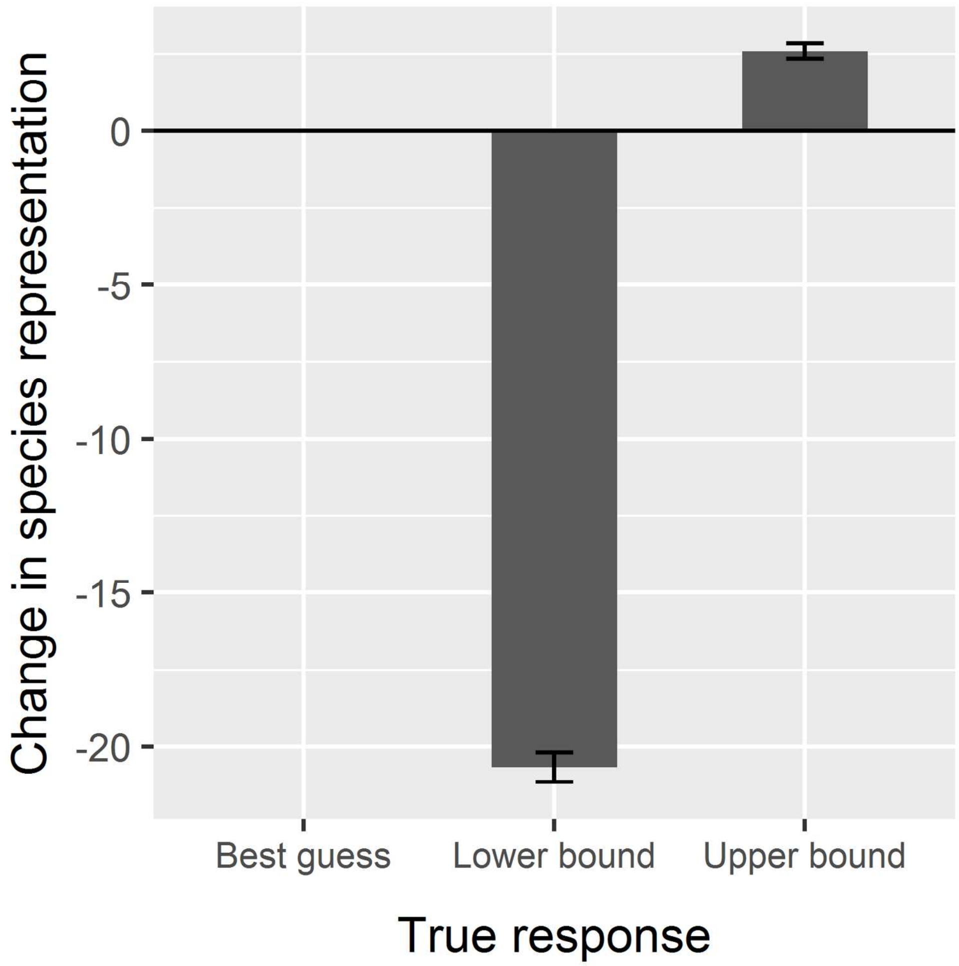

Figure 3. Percentage change in species representation, relative to the best guess scenario, when the true species responses deviate from an expert best guess, due to an expert's own confidence level. The value on the y axis is the percentage change value averaged across species (± 1 SE). Best guess scenario refers to when the best guesses of individual experts (averaged across experts) were assumed to be the true species responses (continuous 0 line). The effect of an expert's own confidence level refers to when the lower and upper bounds of individual experts (averaged across experts) were assumed to be the true species responses. Displayed values are from the run with the lowest objective function, among a set of 10 replicate runs.